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TOWARDS THE "RESILIENCE THINKING": THE EFFECTS OF FOREST MANAGEMENT ON ECOSYSTEM SERVICES PROVISION

SSD: AGR/05 – Forest Management and Silviculture

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[VERSO IL "PENSIERO RESILIENTE": GLI EFFETTI DELLA GESTIONE FORESTALE SULL'APPROVVIGIONAMENTO DEI SERVIZI ECOSISTEMICI]

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Preface

Over the last decades, human-induced effects on Earth systems increasingly undermined the availability of natural capital stocks and flows. Climate change effects, land use and cover transformations, and unsustainable management practices strongly limited the resilience, resistance, and stability of ecosystems, through compromising biodiversity conservation and services provision. In particular, forest resources were largerly threatened or degraded by increasing external disturbances, stresses and impacts, which undermined their capacity to continuously provide benefits to local communities. In order to face these increasing changes, forest management is called to improve ecosystem resilience, mainly through adopting more sustainable strategies.

This research aims at: (i) assessing the effects of management strategies and practices on forest ecosystem resilience, particularly by analyzing and describing the impact of alternative management approaches on biodiversity conservation and services provision; and (ii) providing insights on how to improve forest ecosystem resilience through implementing the "resilience thinking" in practical forest management. The research develops throughout the following steps: (i) the description of how forest management approached the concepts of sustainability and resilience over the last decades, from global to local scale; (ii) the reviewing of the main economic and ecological foundations in assessing forest ecosystem services; (iii) the explanation of some of the most recent approaches for mapping and quantifying forest ecosystem services, including the use of different indicators; (iv) the assessment of the main effects of forest management on ecosystem services provision both at landscape and stand scale; and (v) the delineation of useful guidelines to implement the "resilience thinking" in forest management.

Although the effects from other disturbances (i.e. climate and land use changes) are not treated here, the main research findings may give a substantial contribution to deeper understand the role of forest management in improving forest ecosystem resilience, as well as to better orient adaptive strategies from stand to landscape scale towards ensuring both forest health and vitality and benefits to local communities in the future.

To the three wonderful lights

who look at me from the sky

Main contents

FROM SUSTAINABILITY TO RESILIENCE: A BACKGROUND ON THE EVOLUTION OF FOREST MANAGEMENT 1

1.1 CONTRIBUTION OF FOREST MANAGEMENT TO SUSTAINABLE DEVELOPMENT 2
1.2 THE THREE FORMS OF SUSTAINABLE FOREST MANAGEMENT: INTEGRATED, ADAPTIVE AND ECOSYSTEM-BASED 7
1.3 THE 'RESILIENCE THINKING' IN FOREST MANAGEMENT 10
1.3.1 Ecosystems and resilience: how to face external changes 10
1.3.2 Improving resilience through forest management 12
1.4 TOWARDS THE "RESILIENCE THINKING": CHANGING TRADITIONAL FOREST MANAGEMENT IN ITALY 14
1.4.1 Mediterranean forest landscapes as complex adaptive systems 14
1.4.2 Implementing resilience thinking in forest management in Italy: the systemic silviculture 15
REFERENCES 17

WHAT ABOUT FOREST ECOSYSTEM SERVICES? STATE OF KNOWLEDGE AND FUTURE TRENDS 25

2.1 STATE OF KNOWLEDGE ABOUT FOREST ECOSYSTEM SERVICES 26
2.1.1 Forests and provisioning services 29
2.1.2 Forests and regulating services 30
2.1.3 Forests and biodiversity conservation 34
2.1.4 Forests' amenities, cultural values and recreational activities 35
2.2 ECOLOGICAL AND ECONOMIC FOUNDATIONS IN ASSESSING FOREST ECOSYSTEM SERVICES 37
2.2.1 From biophysical components to services provision 37
2.2.2 Trade-offs and ecosystem services evaluation 39
2.3 CASE STUDY 1: A DOWNSCALED REVIEW ON FOREST ECOSYSTEM SERVICES IN ITALY 42
2.3.1 The context 42
2.3.2 Objectives and methodology 42
2.3.3 Results 44
2.3.4 Discussion and conclusions 49
REFERENCES 53

MODELING FORESTS FOR MULTIPLE SERVICES: ADVANCED APPROACHES AND RECENT TECHNIQUES 66

3.1 TOOLS AND APPROACHES FOR MODELING AND MAPPING FOREST ECOSYSTEM SERVICES 67
3.1.1 The role of decision-support systems in forest management 67
3.1.2 Mapping changes of forest ecosystems and their services 69
3.2 THE INDICATOR-SIDE OF FOREST ECOSYSTEM SERVICES 72
3.2.1 How to measure ecosystem services: towards the use of indicators 72
3.2.2 Building a unified framework of forest ecosystem services indicators 73

3.3 Case study 2: The implementation of a multilevel algorithm to map forest ecosystem functions in a Natura2000 site in Italy 79
3.3.1 The context 79
3.3.2 Objectives and methodology 80
3.3.3 Results 88
3.3.4 Discussion and conclusions 91
3.4 Case study 3: Assessing barriers and drivers for Integrated Forest Management at landscape scale in Italy 94
3.4.1 The context 94
3.4.2 Objectives and methodology 95
3.4.3 Results 100
3.4.4 Discussion and conclusions 104
Acknowledgements 105
References 106

Assessing and monitoring the effects of forest management on ecosystem services provision 121

4.1 Overview on the role of forest management to improve biodiversity conservation and services provision 122
4.2 Case study 4: Managing forests in Protected Areas to maximize biodiversity conservation and services provision 124
4.2.1 The context 124
4.2.2 Objectives and methodology 125
4.2.3 Results 125
4.2.4 Discussion and conclusions 134
Acknowledgements 136
4.3 Case study 5: The effects of forest management on carbon sequestration: the case of mountain Forest Categories in Italy 138
4.3.1 The context 138
4.3.2 Objectives and methodology 139
4.3.3 Results 145
4.3.4 Discussion and conclusions 153
Acknowledgements 157
References 158

Future challenges to implement the "resilience thinking" in forest management 167

5.1 Forest ecosystem complexity, resilience capacity, and management: lessons learned 168
5.2 Understanding the relationships between forest biodiversity and ecosystem services 170
5.1.1 Management strategies to improve resilience in forest stands 172
5.1.2 Management approaches to improve resilience in forest landscapes 174

5.3 Understanding the key-role of community-based management to improve forest ecosystem resilience 178
5.4 Towards the "resilience thinking": key messages 182
References 183

List of Figures **189**

Supplementary material **192**

Appendix 1 193
Appendix 2 211
Appendix 3 221

1

From sustainability to resilience: a background on the evolution of forest management

Earth systems have been strongly modified by human-induced effects, such as climate change, land use change, loss of biodiversity and habitats, reduction of health and stability of ecological assets. Accordingly, this human footprint has been interpreted as the shifting from Holocene to Anthropocene era. Among natural resources, forest ecosystems, as fundamental sources of human benefits, have been increasingly threatened or degraded over the time, especially through unsustainable forest management practices. To face these substantial changes, traditional approaches in forest management (mostly economic-oriented) need to be translated into more sustainable and ecologically-based orientations.

In this chapter, the evolution of forest management thought is described. Evolved so far from the sustainability to the ecosystem-based approach, forest management is nowadays oriented towards improving adaptability, resistance and resilience of forest ecosystems. This shifting of practical forest management objectives is needed to ensure the sustainable use of forest resources, especially to recover degraded landscapes, as in Italy, which suffered the past overexploitation of forest products and the more recent abandonment of traditional forestry practices.

1.1 Contribution of forest management to sustainable development

At the end of the 80's, the integration of sustainability with global development (i.e. economic, social, and environmental), thus drastically changing the previous thinking, was built on several global crises, such as (e.g. Reid 1995): (i) the severe anthropogenic impacts on biogeochemical cycles (e.g. global pollution of atmosphere and oceans; national consumption patterns of fossil fuels contributing to climate change and sea-arise; fresh water pollution; soil degradation and erosion; chemical pollution due to the excessive use of fertilizers and pesticides; soil salinization; etc.); (ii) the reduction of flows of natural and human-made capitals (e.g. loss of biodiversity, habitats integrity and gene-pools; degradation of ecosystems arising from deforestation, fuelwood collection, erosion and urbanization; etc.); (iii) growing inequality between world's rich and poor (i.e. ensuring food access in the face of rising population; the break-down of traditional, ecologically-sound systems of resource management under commercial and population pressures; displacement of economic processes of the resource-poor to marginal lands or to rapidly growth cities, resulting in under-employment; etc.); (iv) powerful trends contributing to the unsustainable development (e.g. industrialization and integration of finance, and marketing and advertising in the global marketplace; the growing aspirations for Western-style consumption patterns fuelled by satellite television; the explosion of capitalist energy in South-eastern Asia and South America; the massive suburbanization in land use, and the expected doubling, by the year 2025, of motor-vehicle numbers; etc.); and (v) issues of governance and mediation in development and the need for long-term, holistic planning (e.g. reconciling market mechanisms and short-term political objectives with longer-term development needs; concerns of international equity among nations to recompense for past resource extraction and pollution; needs to develop decision-making systems and participation mechanisms, which define sustainable processes of development; etc.).

At that time, sustainable development (SD) was defined as: "a development strategy that manages all assets, natural resources, and human resources, as well as financial and physical assets, for increasing long-term wealth and wellbeing; […] a goal that rejects policies and practices supporting current living standards by depleting the productive base, including natural resources, and that leaves future generations with poorer prospects and greater risks than our own" (Repetto 1986; p. 15). SD came into use in policy circles after the Brundtland Commission's report on environment and development in 1987, and was synthetized as "the development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (UN 1987; p. 45). More recently, growing evidence and real-world changes convincingly show that humanity is driving global environmental change and has pushed us into a new geological epoch, the Anthropocene[1], and that as a consequence the SD definitions need to be revised. In particular, Griggs *et al.* (2013) proposed to combine

[1] The concept of the *Anthropocene*, proposed about a decade ago, was introduced to capture trends and variations in biogeochemical cycles in the relationship between humans and the global environment (Crutzen 2002). The term Anthropocene suggests (Steffen *et al.* 2011) that: (i) the Earth is now moving out of its current geological epoch, called the Holocene; and (ii) human activity is largely responsible for this exit from the Holocene, that is, that humankind has become a global geological force in its own right.

the Millennium Development Goals (MDGs) with conditions necessary to assure the stability of Earth's systems into the future, thus creating six Sustainable Development Goals (SDGs), such as: (i) thriving lives and livelihoods; (ii) sustainable food security; (iii) sustainable water security; (iv) universal clean energy; (v) healthy and productive ecosystems; (vi) governance for sustainable societies. Figure 1 summarizes the unified MDGs-SDGs framework.

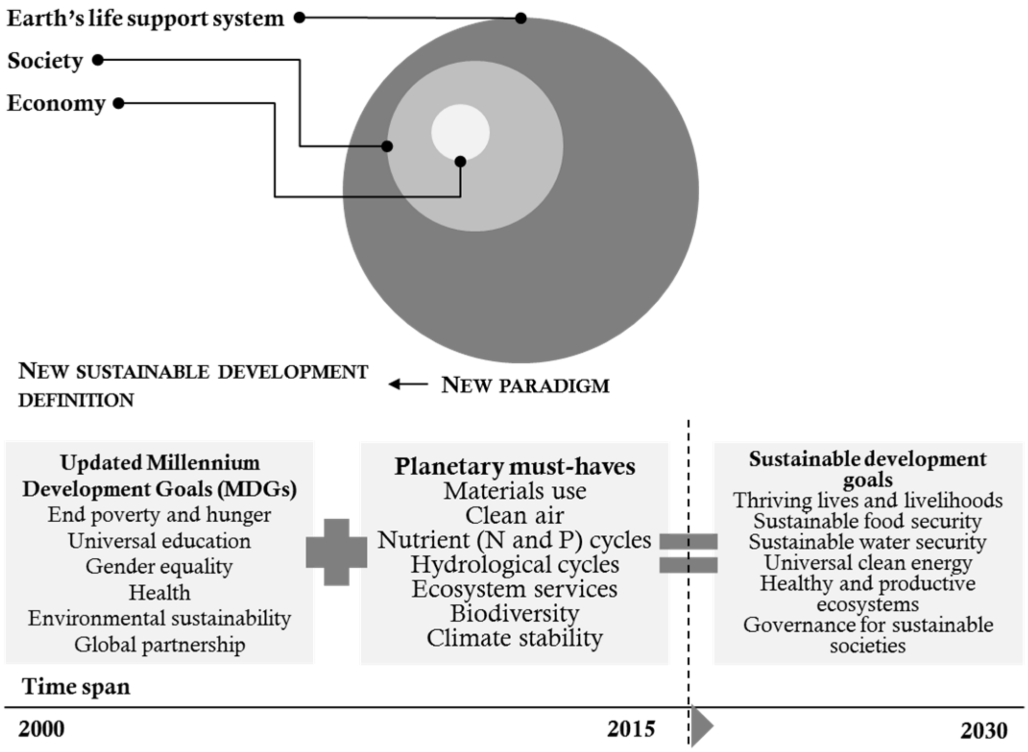

Figure 1: Unified framework for integrating Millennium Development Goals (MDGs) with sustainability conditions in the context of revised SD definition (Griggs *et al.* 2013, modified).

According to such framework, Griggs *et al.* (2013) defined SD as a "development that meets the needs of the present while safeguarding Earth's life-support system, on which the welfare of current and future generations depends".

The key necessary condition for achieving sustainability lies on the constancy of the natural capital stock over the time (Pearce *et al.* 1990). In this way, natural capital properly refers to "a stock that yields a flow of valuable goods and services into the future" and can be differentiated into "renewable natural capital (active and self-maintaining using solar energy) and nonrenewable natural capital (passive)" (Costanza and Daly 1992). Forest ecosystems fall in the first category. It is important to specify that the concept of sustainability has to be implicit within the active natural capital asset, for which any consumption that requires its running down cannot be counted as an income. Considering the natural capital as a whole, forests ecosystems strongly contribute to global social and environmental sustainability by providing simultaneously a wide range of economic, social, environmental and cultural services, despite represent one of the most increasingly overused sources of benefits for humans over the time (Maini 1992). In fact, although forests are recognized as fundamental sources of goods and services on Earth (MEA 2005), they are continuously threatened or degraded by human-induced effects (Foley *et al.* 2005), such as the global climate change (Lindner *et al.* 2010), land use and cover change (Deng *et al.* 2013), and unsustainable management practices (Haberl *et al.* 2007).

Evolved from the concepts of both sustained yield and sustainable forestry over the last decades, Sustainable Forest Management (SFM) is an attempt to maintain the flows of different sets of goods and services, by considering forests as integrated parts of dynamic landscapes (Sayer and Maginnis 2005). A set of principles to underlie SFM worldwide were discussed and approved during the United Nations Conference on Environment and Development (UNCED) in Rio de Janeiro in 1992 (i.e. The Statements of Forest Principles, SFP). SFP can be summarized as follows: (i) all countries, notably developed countries, should make an effort to 'green the world' through reforestation and forest conservation; (ii) States have a right to develop forests according to their socio-economic needs, in keeping with national sustainable development policies; and (iii) specific financial resources should be provided to develop programmes that encourage economic and social substitution policies. To date, many initiatives have sought to develop tools supporting SFM (Hickey *et al.* 2005). Since UNCED, international progress has been made in a variety of ways to adopt and implement the SFM concept in policy-making processes from the continental to the forest unit level (Wijewardana 2008). The International Tropical Timber Organisation (ITTO) introduced the Criteria and Indicators concept and terminology in 1992 (ITTO 2005). Since then several regional groupings of countries have worked together upon the process of generating and testing appropriate C&I to suit their own conditions (Prabhu *et al.* 1998). In 1993 thirty-eight European countries signed on to the temperate forest 'Helsinki process' (MCPFE 1993) and twelve non-European countries, also with temperate forests, followed suit through the 'Montreal process' (Canadian Forest Service 1995). In 1995, eight Amazonian countries began work on the 'Tarapoto process' (Elías 2004) and more recently twenty-seven sub-Saharan African countries have been working on Criteria and Indicators (C&I) for dry zones. Processes are under way in the near East and Central America, and recently most of all the African Timber Organisation has been testing C&I for the rainforest zones of Central and West Africa (Castañeda 2000).

As already mentioned, at European level the Ministerial Conference on Protection of Forests in Europe (MCPFE, now ForestEurope) adopted the following SFM definition: "the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality, and their potential to fulfill, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems" (MCPFE 1993). Subsequently, various guidelines for correctly implementing SFM have been developed (MCPFE 2003).

Over the past 25 years, the framework of SFM-C&I has been developed as a powerful tool for implementing SFM (EFI 2013). Table 1 reports the updated list of SFM-C&I as developed by MCPFE.

Table 1: Improved pan-European criteria and indicators for sustainable forest management (MCPFE 2003).

Criterion	Indicator
C1 – Maintenance and appropriate enhancement of forest resources and their contribution to global carbon cycles	1.1 – Forest area and other wooded land (OWL) 1.2 – Age structure and/or diameter distribution 1.3 – Growing stock 1.4 – Carbon stock
C2 – Maintenance of forest ecosystem health and vitality	2.1 – Deposition and air pollutants 2.2 – Soil conditions 2.3 – Defoliation 2.4 – Forest damage
C3 – Maintenance and encouragement of productive functions of forests	3.1 – Increment and fellings 3.2 – Roundwood 3.3 – Non-wood goods 3.4 – Services 3.5 – Forests under management plans
C4 – Maintenance and appropriate enhancement of biological diversity in forest ecosystems	4.1 – Tree species composition 4.2 – Regeneration 4.3 – Naturalness 4.4 – Introduced tree species 4.5 – Deadwood 4.6 – Genetic resources 4.7 – Landscape pattern 4.8 – Threatened forest species 4.9 – Protected forests
C5 – Maintenance and appropriate enhancement of protective functions of forests	5.1 – Protective forests (soil, water, and other ecosystem functions) 5.2 – Protective forests (infrastructures and managed natural resources)
C6 – Maintenance of other socio-economic functions and conditions	6.1 – Forest holdings 6.2 – Contribution of forests to the Gross Domestic Product (GDP) 6.3 – Net revenue 6.4 – Expenditures for services 6.5 – Forest sector workforce 6.6 – Occupational safety and health 6.7 – Wood consumption 6.8 – Trade in wood 6.9 – Energy from wood resources 6.10 – Accessibility for recreation 6.11 – Cultural and spiritual values
A – Overall policies, institutions and instruments for sustainable forest management	A.1 – National forest programmes or similar A.2 – Institutional frameworks A.3 – Legal/regulatory frameworks and international commitments A.4 – Financial instruments/economic policy A.5 – Informational means
B – Policy, institutions and instruments by policy area	B.1 – Land use and forest area and OWL B.2 – Carbon balance B.3 – Health and vitality B.4 – Production and use of wood B.5 – Production and use of non-wood goods and services, provision of especially recreation B.6 – Biodiversity B.7 – Protective forests

Criterion	Indicator
	B.8 – Economic viability B.9 – Employment (incl. safety and health) B.10 – Public awareness and participation B.11 – Research, training and education B.12 – Cultural and spiritual values

Literature is mounting worldwide about the state of the art, challenges and opportunities of the SFM implementation. Outside Europe, many SFM-related studies are available from Northern America (e.g. Riley 1995), Southern America (e.g. Pokorny and Adams 2003), South-eastern Asia (e.g. Muhammed *et al.* 2008), Australia (e.g. Howell *et al.* 2008), and Africa (e.g. Kruger and Everard 1997).

To date, the main merits of the implementation of SFM-C&I have been (Grainger 2012; Wijewardana 2008): (i) support a global understanding of what constitutes SFM; (ii) foster political processes on SFM; (iii) find a common symbolic language to overcome historic conflicts (e.g., forestry vs. environmentalists) and hence support consensus-finding; (iv) find a common terminology in the global environmental governance; (v) streamlining and structuring forest reporting; (vi) support unambiguous communication and learning among stakeholders; and (vii) serving as a means for education and capacity-building by fostering participatory decision-making and decentralized policy implementation. However, some general limitations are evident, as follows: (i) unbalanced indicator sets, which are particular weak in socio-economic indicators (Gough *et al.* 2008); (ii) harmonization, terms and definition on forest information is still imperfect and hampers reliable C&I interpretation (Irland 2010); (iii) monitoring and streamlined reporting are still challenges for policy makers and forest managers (Hickey 2008); (iv) C&I are strongly outcome-centered measures but fail in identifying direct links to and evidence on forest management activities and responses (Foster *et al.* 2010); (v) C&I do not consider linkages, interdependencies, and causal chains among indicators (Requardt 2007), as well as do not connect quantitative and qualitative policy indicators; and (vi) C&I fail to facilitate more systemic analysis of how SFM is embedded in socio-ecological systems (Wolfslehner and Vacik 2011).

A key issue in the future development of C&I refers to maintain an active link between research efforts and operational needs to ensure the best use of resources and timely solutions (Wijewardana 2008). More in general, the fundamental question of SFM lies in integrating levels of response and identifying linkages among the various pieces on the forest landscape (Wang 2004). As a consequence, enhanced SFM requires better reporting and verification, more areas covered and enhanced implementation of C&I. Further progress in improving forest management worldwide also relies on gathering better information needed to monitor and analyze global forest trends. A mix of effective public policies and private markets continues to be needed to help achieve global SFM (Siry *et al.* 2005).

1.2 The three forms of sustainable forest management: integrated, adaptive and ecosystem-based

As already described, SFM changed the traditional forest management approach, which was primarily oriented to maximize timber provision and related economic incomes (Puettmann *et al.* 2009). Without considering the efforts to implement SFM into practice over the time, further changes in forest management approach are needed, because current intensive management may potentially undermine the capacity of forests to sustain the flow of benefits for humans in the future (Bennett and Balvanera 2007; Fischer *et al.* 2009), especially considering the current contexts of climate change and anthropogenic alterations of biogeochemical cycles (e.g. Dale *et al.* 2001; Allen *et al.* 2010). Sayer and Maginnis (2005) suggested that some of the most important underlying trends that created the need for more integrated and holistic forest management strategies are: (i) broadening forest management objectives (at all scales, forest owners and managers have to deal with a much broader range of social and environmental issues than in the past); (ii) codifying good practice (regulators, certifiers, and civil society are developing several performance indicators to assess the efficiency of forest management and the health of forests); (iii) recognition of pluralism in forest management (recognizing that there are many different systems of ownership and use of forests that are likely to be sustainable); (iv) decentralization-devolution; (v) globalization (multi-national corporations, banks, trade regulations etc. have strong impacts on how forests are managed and are usually out of local control); (vi) climate change (the uncertainties created by the potential impacts of different climate change scenarios have major implications for forest conservation and management laws and institutions; see also Millar *et al.* 2007); and (vii) governance (forests are only well managed when formal institutions are effective and civil society is mobilized to defend the interests of diverse stakeholders; see also Armitage 2005).

In the SFM debate, the concepts of eco-regions, landscape suitability and functionality, integration of conservation and development have led to three main SFM integrated implementation characters (see e.g. Döbert *et al.* 2014), such as: (i) Integrated Forest Management (IFM); (ii) Ecosystem Approach (EA); and (iii) Adaptive Forest Management (AFM). Generally, these three concepts require that forest management and planning have to consider in practice (i) the adaptation of forest ecosystems to the changing environment, (ii) the relationships of forests with other neighboring ecosystems and/or land uses, and (iii) the impacts of socio-economic elements on health, stability and regeneration capacity of forest ecosystems. Figure 2 reports the IFM, EA, and AFM definitions.

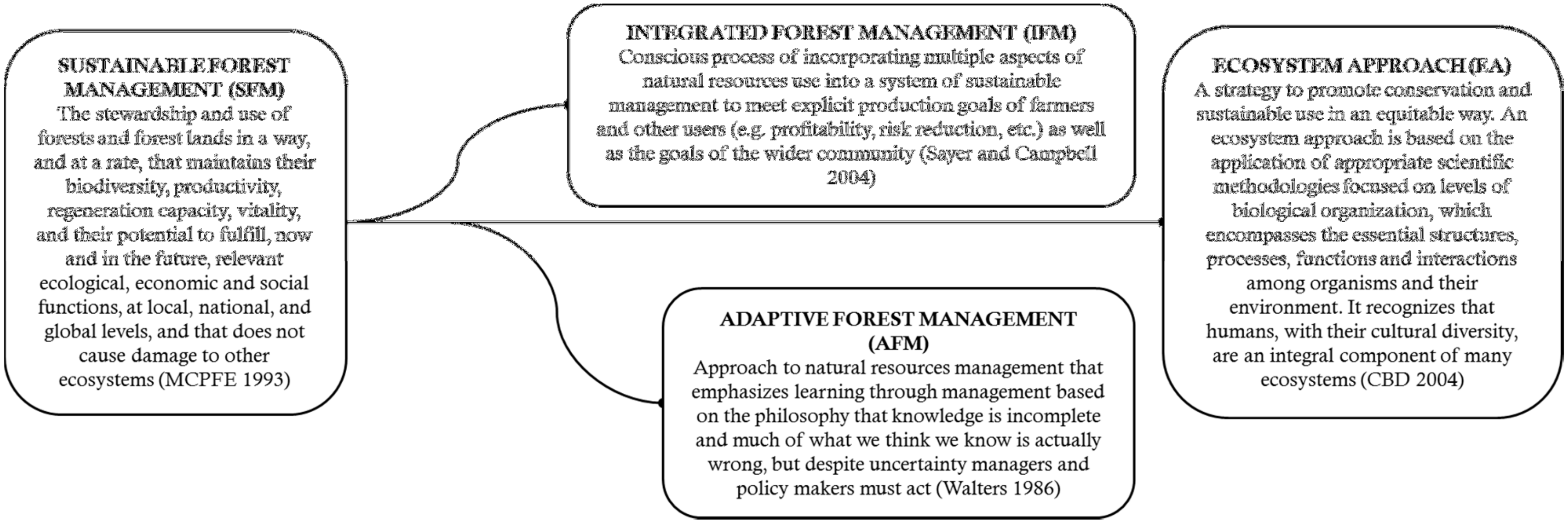

Figure 2: Evolutionary flowchart reporting the key definitions of Integrated Forest Management, Ecosystem-based Management, and Adaptive Forest Management as means to maintain sustainability of forest management.

While SFM and EA can be though as similar responses to the same set of underlying driving forces, there are important differences in the origins and philosophies of the two concepts. SFM has been developed and debated by forestry professionals, with their primary focus on obtaining goods and services from the land under their control. By the other hand, the EA debate has been led by a more heterogeneous group of proponents more concerned with conservation. Indeed, as it has developed, EA represents a compromise between a rich country 'precautionary' agenda and a developing country 'development' agenda where poverty reduction and economic growth are predominant concerns. The SFM approach is built on the progressive evolution of the sustained yield and forestry notions, towards sustaining the flows of different goods and services over the time. In the same way, EA is a tool to promote sustainability in natural resources management by incorporating a broader set of management and participation objectives (CBD 2004).

By integrating sustainable forestry with ecosystem-based approach, ten principles have to be respected (*sensu* Sayer and Maginnis 2005): (i) there is no single management of ecosystems, but multiple approaches need to be adapted and applied pragmatically in each situation; (ii) people are integrated part of the whole environment; (iii) the management approach is based on experimenting treatments, learning from the environmental responses, and implementing decisions, accordingly; (iv) land rights, policy regulations, and forestry law are as important as practical management, because they enhance and enable the best practices and trajectories; (v) science does not provide answers, but it helps land managers to learn from mistakes, adapt and explore innovative options; (vi) EA and SFM require tools that measure the performance of the whole system, and are conceived to reduce the gap between managers, stakeholders and decision-makers, thus ensuring that people and environment can live together without any misbalanced decline.

1.3 The 'resilience thinking' in forest management

1.3.1 Ecosystems and resilience: how to face external changes

Physical influence on ecosystems include geology, climate, topography, hydrology, connectivity with other ecosystems, and the results of human activity (Elmqvist *et al.* 2010). Larger disturbances (such as e.g. anthropogenic eutrophication, toxic pollution, habitat loss, disconnection from adjacent ecosystems, species invasion, climate change, etc.) can drive permanent or long-term ecosystem changes by altering the physical structure of ecosystems, and through removal of species and alteration of species interactions (see e.g. Ellis *et al.* 2013). The capacity of an ecosystem to withstand perturbations without losing any of its functional properties is often referred to as ecosystem resilience (see §2.1, box 1). Walker *et al.* (2004) defined ecosystem resilience as its capacity to absorb disturbances and reorganize while undergoing change, thus retaining essentially the same function, structure, identity and feedbacks. Accordingly, the 'resilience thinking' describes the collective use of a group of concepts to address the dynamics and development of complex social-ecological systems; resilience, adaptability and transformability are central (Folke *et al.* 2010). In general, 'resilience thinking' embraces a collection of ideas and theories that have become widely applied to individual case studies, for example ideas such as regime shifts, thresholds, transformation, adaptive cycles and social-ecological systems (Rist and Moen 2013).

Where environmental drivers are persistent or strong, ecosystems may pass a threshold and undergo sudden and catastrophic structural change (Thom 1969; Jørgensen 1997; Walker and Myers 2004). This can shift the ecosystem to an alternative state (Holling 1973; May 1977; Sheffer *et al.* 2001), which is also termed a 'regime shift' (Folke *et al.* 2004), and cause profound changes in ecosystem services, biodiversity and aesthetics values (Sheffer *et al.* 2001). Although the causes for a 'regime shift' may be ascribed to a recent short-term event (i.e. exceptionally dry periods), a deeper analysis shows interacting causal networks of slow and fast processes that have eroded the resilience of a system, thereby making it more vulnerable to shocks and disturbances (Hughes *et al.* 2013; see Figure 3). Among the most important drivers for regime shifts (Patz *et al.* 2005), those strictly related to forestry field can be: (i) the wildlife habitat destruction, conversion or encroachment, particularly through deforestation and reforestation; (ii) the biological asset change (including loss of predator species and changes in host population density); and (iii) agricultural land use changes, and climate variability and change. In order to globally reduce the 'regime shifts', Rockström *et al.* (2009) conceptualized the 'planetary boundaries' for estimating the safe-operating space for humanity with respect to the functioning of the Earth systems. Accordingly, the erosion of resilience manifests itself when long periods of seemingly stable conditions are followed by periods of abrupt, non-linear changes, reflected in critical transitions from one stability domain to another when thresholds (intrinsic features defined by control variables, such as temperature and the ice-albedo feedback in the case of sea ice are crossed; Scheffer *et al.* 2001; Walker *et al.* 2004; Lenton *et al.* 2008).

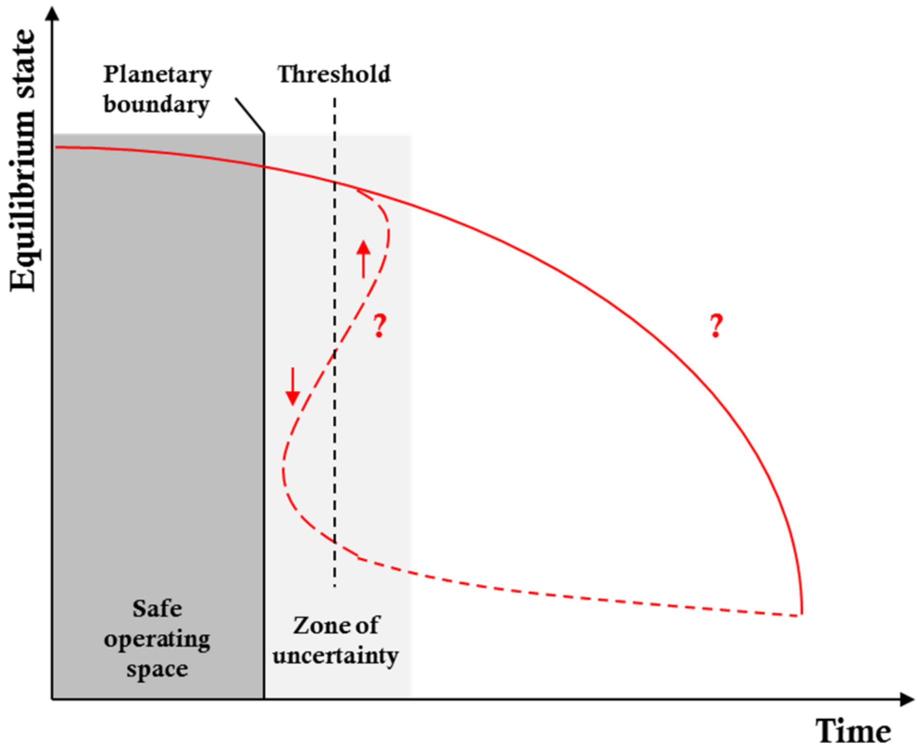

Figure 3: Conceptual description of planetary boundaries. The equilibrium response is plotted as a function of the strength of multiple, interacting anthropogenic drivers, such as overharvesting or ocean acidification. Uncertainty over the eventual impact of high levels of drivers is indicated by considering two potential system responses at equilibrium: smooth, and hysteretic (or folded). The latter constitutes threshold effects (Hughes *et al.* 2013, modified).

In conjunction with climate change and land-system change, one of the most dangerous influence on Earth systems functioning refers to the rate of biodiversity loss. Indeed, current and projected rates of biodiversity loss constitute the sixth major extinction event in the history of life on Earth – the first to be driven specifically by the impacts of human activities on the planet (Chapin III *et al.* 2000). Ecosystems (both managed and unmanaged) with low levels of response diversity with functional groups are particularly vulnerable to disturbances (such as disease) and have a greater risk of undergoing catastrophic regime shifts (Sheffer and Carpenter 2003). For example, Rockström *et al.* (2009) suggested <10 Extinctions per Million Species per Year (E MSY^{-1}) as a safe planetary boundary for the rate of biodiversity loss. This is clearly being exceeded by at least one to two orders of magnitude, indicating an urgent need to radically reduce biodiversity loss rates (Díaz *et al.* 2005).

In predicting regime shifts, Pardini *et al.* (2010) suggested that future research should aim to test (i) the effectiveness of restoring native vegetation cover across different landscape contexts, and (ii) the possible long-term consequences of large-scale shifts in biodiversity for ecosystem functions and services. Therefore, avoiding detrimental consequences of planetary-scale regime shifts will require a clear focus on the drivers and feedbacks, not just on disconnected efforts to control some of the biological consequences (Hughes *et al.* 2013). Further efforts are needed to identify Earth system thresholds, as well as apply precautionary principles, upon which current governance and management approaches are often lacking to act (see e.g. Walker *et al.* 2009).

1.3.2 Improving resilience through forest management

Resilience of forest ecosystems is increasingly influenced by anthropogenic changes and threatened by abiotic perturbations (e.g. Folke *et al.* 2004). From global to local scale, major impacts originate from climate change (Dale *et al.* 2001), atmospheric pollution (Gundersen *et al.* 2006), land use transformations (Kulakowski *et al.* 2011), biodiversity loss and habitat fragmentation (Isbell *et al.* 2013), wildfires (Churchill *et al.* 2013), insects outbreaks (Seidl *et al.* 2011), storms, floods or avalanches, overgrazing, and forestry activities (leading to large deforestation or afforestation processes) (Franklin *et al.* 2007). The combination of these disturbances can reduce the forest ecosystem functioning and a sustainable goods and services provision in the future (e.g. Toman and Ashton 1996; Costanza *et al.* 2000). Rather than specific and sustained targets, such as allowable annual cuts or a minimum amount of wildlife habitats, forest management should be oriented to enhance the resilience of ecosystem states[2] deemed essential to the provision of ecological goods and services while at the same time decreasing the resilience of states that do not provide these or that do so at low levels (Gunderson and Holling 2004). Among the others, forest structural diversity, measured as variation along a vertical or horizontal profile, appears a good indicator of forest management effects on ecosystem resilience (Roberts and Gilliam 1995; Lindgren and Sullivan 2001). In addition, there is increasing evidence that spatial heterogeneity at multiple scales is a critical component of ecosystem resilience (Levin 1998; Moritz *et al.* 2011; North *et al.* 2009; Stephens *et al.* 2008).

Forest management is able to implement "resilience thinking" in practice, only if forest ecosystems are considered as complex adaptive systems, for the following reasons (Puettmann *et al.* 2013): (i) the shift from dominance of a single management objective (i.e. wood production) towards the provision of multiple objectives opens the door to a less controlled and focused paradigm; and (ii) increased future uncertainty due to future external perturbations (i.e. climate change) requires a more flexible management approach. Thinking of forests as complex systems is a relatively recent development in the fields of ecology (e.g. Levin 2005) and forest management (Campbell *et al.* 2009). Forests exhibit all characteristics of complex adaptive systems (Chapin III *et al.* 2009): they are heterogeneous, highly dynamic, and contain many biotic and abiotic elements which interact across different levels of organizations with various feedback loops (Puettmann *et al.* 2013). Managing forests as complex systems requires (i) a stronger emphasis on multiple temporal, spatial and hierarchical scales, (ii) more explicitly considering interactions among multiple biotic and abiotic components of forests, (iii) understanding and expecting non-linear responses, and (iv) planning for greater uncertainty in future conditions (Puettmann *et al.* 2013). At European level, Bengtsson *et al.* (2000) argued that the next generation of forestry practices would need to: (i) deeper understand natural forest dynamics; (ii) analyze the role of biodiversity (i.e. key species and functional groups) in affecting the ecosystem functionality; and (iii) implement and adapt prescriptions in accordance with natural dynamics; (iv) result from an interaction between ecology, forestry,

[2] To estimate resilience, it is necessary to specify the state(s) and spatial scale of the ecosystem being considered (resilience *of* what), the perturbations of interest that affect the persistence of system states (resilience *to* what), and the temporal scale of interest (Carpenter *et al.* 2001; Walker and Meyers 2004).

economy, and social fields in order to establish a value of the important goods and services from forest ecosystems. Table 2 highlights the main differences between the resilience thinking, the ecosystem approach and the adaptive management.

Table 2: Comparison of management paradigm characteristics (Rist and Moen 2013, modified).

	Resilience thinking	Ecosystem approach	Adaptive management
Management goal	To maintain the system identity – system function, structure and feedbacks (Walker and Salt 2012). To maintain a desirable state (identity), or transform into a more desirable state (Walker *et al.* 2006)	To promote ecological integrity while allowing human use on a sustainable and equitable basis (CBD 2013a).	To produce updated understanding as well as economic product (Walters and Holling 1990). Additional aims include participation and improved decision making (Holling 1978)
Links to complexity (i.e. system dynamics)	Considers multiple local but no global equilibrium, hysteresis, alternative stable states, regime shifts and transformation are key concepts (Gunderson 2000; Walker and Salt 2012)	Non-linearities are recognized but not non-equilibrium dynamics, the perspective taken is akin to engineering resilience (CBD 2013a)	Non-linearities and non-equilibrium dynamics are recognized (Holling 1978; Walters 1986) but there an assumption of dynamic stability in the underlying environment such that learning is possible (Holling 1978; Walters 1986)
Management implementation tools	Resilience assessment (Resilience Alliance 2007; 2010), Adaptive co-management (Folke *et al.* 2005; Olsson *et al.* 2004; Resilience Alliance 2013), adaptive Management and governance (Walker and Salt 2012)	Translation into practice via principles and operational guidance (CBD 2013b; Maltby 2003)	Modelling including systems modelling techniques as well as scientific methods of data collection, assessment and evaluation. Workshop processes are also a key feature (Holling 1978; Lee 1994)
Management performance evaluation tools	Specified resilience can be measured by surrogates or indicators (e.g. Allen *et al.* 2005; Bennett *et al.* 2005). General resilience is indicated by diversity, modularity, tightness of feedbacks, openness and reserves (Walker and Salt 2012)	Ecosystem health e.g. biological diversity, food chain characteristics, ecosystem productivity or ecosystem functions (CBD 2013b)	Metrics are context specific but focus on system components rather than processes (Walters and Holling 1990)

To improve forest ecosystem resilience, the objective is to understand natural processes and resultant patterns and draw upon this understanding to design silvicultural approaches that achieve ecological and other management goals (e.g. Franklin *et al.* 2007). To meet the above-mentioned emerging challenges, as well as to face external changes and disturbances, forest management is called to combine old and new tools and skills, thus enabling a more flexible, adaptive and experimental approach (e.g. Puettmann *et al.* 2013). Accordingly, the major questions that future research needs to address include (Elmqvist *et al.* 2010): (i) understanding the links between biodiversity, ecosystems and resilience; (ii) understanding the dynamics of ecosystem services; and (iii) understanding the dynamics of governance and management of ecosystems and ecosystem services.

1.4 Towards the "resilience thinking": changing traditional forest management in Italy

1.4.1 Mediterranean forest landscapes as complex adaptive systems

Over the history, the Mediterranean basin has experienced the birth, blooming and collapse of the largest and powerful civilizations in the world (Blondel 2006). The use of forests in Mediterranean Countries has been characterized by the following subsequent processes (Nocentini and Coll 2013): (i) first hunters/gatherers collecting wood and other forest products; (ii) shaping of varied sustainable agro-pastoral-systems connected to local traditions and economies; (iii) more recent consequences of industrial development and applications of scientific forest academic principles; (iv) large reforestation processes which were carried out in many areas to heal the wounds of excessive exploitation; and (v) present day dichotomous situation of land abandonment and localized over-exploitation.

Taking into account the historical relationships between humans and forest landscapes (Scarascia-Mugnozza *et al.* 2000), the Mediterranean basin (and its diversification in space and time) can be treated as a complex adaptive system. In Mediterranean forests, feedback loops of biotic and abiotic interactions across hierarchical scales create persistent and structures and scale-specific patterns (Allen and Holling 2010). Much of these positive and negative feedback loops (involving humans and operating for long time periods) was through trial and error, and can be easily identified in traditional silviculture and agroforestry systems (Blondel 2006). In particular, forest management, which traditionally provided a great diversity of products, have slowly focused towards the almost exclusive wood production, thus resulting in a repeated over-simplification of forest stands (e.g. extensive coppice forests or even-aged pure stands; Ciancio and Nocentini 2000). In Italy, forests that have not been directly affected by human uses are found in unique conditions, such as e.g. remote areas or areas for protection purposes (i.e. against avalanches and landslides), and generally show characteristics of 'old-growthness' (Piovesan *et al.* 2005; Burrascano *et al.* 2008; Burrascano *et al.* 2009). In Italy, at least three peculiar features describing human-forest interactions can be identified, as follows: (i) the widespread of coppice forests with standards (in many cases pure stands), which is related to firewood and charcoal production (Ciancio and Nocentini 2004); (ii) the soil fertility degradation and the increasing instability of slopes, due to approximately 1.3 million ha of coniferous plantations (e.g. stone pine [*Pinus pinea* L.], black pine [*Pinus nigra* Arn.], and Calabrian black pine [*Pinus nigra sub. laricio* Poir.], Atlas cedar [*Cedrus atlantica* Man.], radiata pine [*Pinus radiate* Don.] and Rocky mountain douglas fir [*Pseudotsuga menziesi* Mirb.]) started at the beginning of the last century (Corona *et al.* 2009); and (iii) the perpetuation of forest fires, due to the successful establishment (adoption of regeneration strategies) of Mediterranean pines (i.e. *Pinus pinaster* Ait. and *Pinus halepensis* Mill.).

From larger to smaller scale, Mediterranean landscapes and related forest stands have become simplified, due to either intensification of production systems (both agricultural crops and woods) or as a consequence of land abandonment, encroachment and desertification. As a consequence, remaining challenges for Mediterranean forests refer to (i) maintaining diverse traditional forest landscapes, mainly because they offer more options to face future changes, and (ii) increasing heterogeneity and adaptability of simplified forest systems under changing conditions.

1.4.2 Implementing resilience thinking in forest management in Italy: the systemic silviculture

In Italy, climate and environmental modifications, economic development, and population growth have been the main drivers of forest landscape changes over the last century. These trends firstly led to an over-simplification of forest stands and to a widespread of more focused forest management systems, and recently to a net forest area gain originating from the abandonment of rural areas and traditional practices. Recovering biodiversity loss, as well as the health and stability of forest ecosystems, requires a fundamental change in traditional forest management approaches and silvicultual practices, in order to improve the resilience and adaptability of degraded forests to increasing external changes. Over the past, Classical forest management approaches treated population and ecosystem dynamics as if they were acting in a stable environment and according to predictable trajectories (i.e. a top-down control of natural processes). Therefore, classical silvicultural schemes aiming at maintaining specific forest structures and optimizing timber yields were criticized as inappropriate for the complex non-linearity of forest ecosystems.

Since 1990s, the concept of systemic silviculture has been increasingly recognized as a set of methods and operational procedures that are consistent with many attributes of complex systems and complexity science (Ciancio and Nocentini 1997, 2000, 2011; Ciancio *et al.* 2003). Systemic silviculture is defined as "an experimental science based on the study, cultivation and use of the forest, [that is] an extremely complex system […] capable of self-perpetuation and of accomplishing of multiple functions" (Ciancio 2011). The overall aim is to maximize the use of internal energy within the forest to achieve our forest management objectives, instead of relying on external produced energy inputs (Allen and Hoekstra 1992). In other words, the systemic silviculture orients forest management towards the re-naturalization of simplified forests to foster rehabilitation of natural processes: natural self-regulating and self-perpetuating mechanisms that increase a system's resistance, resilience and adaptability (Ciancio and Nocentini 1997, 2011; Ciancio *et al.* 2003). Hence, forest management moves from approaches based on forecasting and anticipating (i.e. the basis of anticipatory management) to approaches based on monitoring the effects interventions on stand growth and development over the time (Ciancio and Nocentini 2004; Corona and Scotti 2011). Table 3 shows a comparison between the classical and the systemic silviculture approach.

Table 3: Comparison between classical and systemic silviculture and management (Ciancio and Nocentini 2011, modified).

	Classical silviculture and management	Systemic silviculture and management
Stand structure	Predetermined stand structure	Unstructured forest (stand structure undefined in space and time)
Species composition	Selected species according to management objectives	Spontaneous species mixture
Silvicultural treatment	Predefined silvicultural treatment	Cautious, continuous and capillary interventions
Cultivation cycle	Predefined rotation period	Undefined
Forest management model	Standard (theoretical) forest management system	Self-organization of forest (monitoring and adaptation to evolutionary processes)
Production	Constant and maximum annual harvesting rate	Periodic harvesting rate
Control	Centralized control according to revenue and market trends	Decentralized control (local knowledge and needs)
Biological concerns	Simplified	Diversified

In practice, systemic silviculture is implemented through the following forest management approaches (Ciancio and Nocentini 2011): (i) the abandonment of rigid schemes; (ii) the encouragement of natural regeneration processes; (iii) the minimization of risk for reducing biodiversity; and (iv) the reduction of nutrient cycles alterations. More generally, enhancing the forest ability to self-organize can be accomplished through implementing multi-objective thinning treatments and other forestry interventions oriented to (Nocentini and Coll 2013): (i) increase the vigor of remaining trees (i.e. resistance to change); (ii) encourage the development of understory vegetation and advanced regeneration (i.e. increasing adaptability); (iii) promote the establishment of drought-tolerant plants and assist transitions to plant communities which are more adaptable to climate change conditions; (iv) enhance the response-type diversity of the system (*sensu* Puettmann 2011); and (v) creating a range of stand structures representing different developmental stages and thus decrease vulnerability to perturbations. By applying the "resilience thinking", not only in Italy, for example Fabbio *et al.* (2003) suggested to better understand the forest ecosystems functioning and dynamics, as well as to concentrate restoration efforts in the most sensitive and naturally not recoverable situations. In addition, forestry research should support practical forest management in the following ways (Scarascia-Mugnozza *et al.* 2000): (i) by studying the effects of landscape structure on ecosystem functioning and resilience, in relation to natural and human-made disturbances; and (ii) by linking the analysis of human-forest interactions to the transfer of knowledge to landscape/forest managers and decision-makers at all hierarchical levels.

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2

What about forest ecosystem services? State of knowledge and future trends

Forests represent fundamental sources of ecosystem services on Earth. Improving the most balanced set of ecosystem services would require an exhaustive understanding of ecosystem functioning, in terms of resistance, stability and resilience characteristics. During the last two decades, increasing research contributions largely provided the ecological and economic bases to integrate the ecosystem services-related concepts with management objectives, but more efforts are still needed to implement them into practice. Therefore, current research is still lacking of more detailed knowledge about how to consider ecosystem services in supporting forest management and decision-making processes, especially at national level.

This chapter offers an overview on the importance of forest ecosystem services, from global to national level. At first, the current state-of-knowledge is unrevealed, by deeply analyzing the forest ecosystem services categories. Secondly, the biophysical and economic foundations of assessing forest ecosystem services are provided. Finally, a downscaled review on the state of the art about forest ecosystem services knowledge in Italy is reported. Main results explain that biodiversity and ecosystem services fluxes have to be further analyzed, and that the linkages between forest managers, stakeholders, and decision-makers have to be enhanced for improving the forest ecosystem services provision.

2.1 State of knowledge about forest ecosystem services

Forests represent extremely important sources of ecosystem services (ES) on Earth. The Millennium Ecosystem Assessment pointed out that forests (MEA 2005): (i) are important refuges for terrestrial biodiversity as well as provide habitat for half or more of the world's known plant and animal species; (ii) play a significant role in global carbon cycle and, consequently, in mitigating the global climate change; (iii) provide a large set of products (timber and non-timber) that are needful for human well-being and livelihood of rural and poor populations; (iv) host catchments to provide accessible and fresh water (both in quality and quantity); and (v) defend cultural, spiritual, and recreational values for many societies.

Research focusing on the links between forest ecosystems and their services has not long tradition. Martínez *et al.* (2009) explored the impact of land use change in terms of ES provision by following two approaches, one focused on hydrological services and another one focused at a larger scale and referred to the Ecosystem Service Value (ESV) of several ecosystems and their services at the same time. In the same way, Fu *et al.* (2013) calculated the monetary value of flood mitigation service by formulating an exponential function with respect to the amount of storm and flood damage according to the Soil Conservation Service (SCS) method, and Band *et al.* (2012) extended an eco-hydrological modeling approach to include hydrologic and canopy structural pattern impacts on slope stability, with explicit feedbacks between ecosystem water, carbon and nutrient cycling, and the transient development of landslide potential in steep forested landscapes.

On the other hand, a large part of publications concerning forest ecosystem services (FES) is focused on policy measures and decision-making processes-related issues to improve and enhance the availability of FES at different scales, from landscape to global level. Deal *et al.* (2012) outlined some of the policy and regulatory frameworks for some of the emerging markets for ES in United States (US), and discussed the role that different regulatory agencies play for each of these services. According to the experiences made and lessons learned from the implementation of many large-scale ES policies in China, Liu *et al.* (2013) pointed out some suggestions on how to improve their effectiveness, efficiency, and sustainability. Molnar and Kubiszewski (2012) described some examples of programs (US and Canada) seeking to maintain ES from wetlands, agricultural lands, forests and water quality, thus suggesting that new policies are necessary to implement the ES values into broader economic decisions. Grêt-Regamey *et al.* (2012) presented a new approach for mapping the uncertainties in the assessment of multiple ES, thus demonstrating that this approach can provide key information for decision-makers seeking critical areas in the delivery of ES in a case study in Swiss Alps. The correlations between the economic values of FES (i.e. tradeoffs), the economic benefits of FES provisioning, and the forest management were treated by several authors (see Gren and Isacs 2009; Holl and Aide 2011; Dymond *et al.* 2012; Ojea *et al.* 2012b).

Many interesting contributions about FES concern the role of forest planning in biodiversity and habitat integrity/ecosystem functionality conservation (e.g. Prato 2009; DeClerck *et al.* 2010; Freudenberger *et al.* 2012; Onaindia *et al.* 2013). Papers focusing on both forest ecosystem processes and on a specific service are poorly available. Maes *et al.* (2013a) studied the recreation opportunity spectrum approach as a useful method to identify areas in terms of their accessibility and potential to provide recreation services, as well as they demonstrated that available data are sufficient to map the potential of ecosystems to provide pollination services. Hanson *et al.* (2013) applied the Habitat Equivalency Analysis (HEA) approach to quantify forest fire habitat damages. They reviewed and identified critical issues that may affect the estimate of lost services following high-severity fires, including potential approaches for dealing with uncertainty. Willaarts *et al.* (2012) presented an innovative method to empirically assess the underlying relationship between the use and management of Mediterranean agro-ecosystems, their spatial pattern of green and blue freshwater flow generation and the provision of hydrologic ES (HES), through using the BalanceMED model. Modeling FES to describe landscape spatial characteristics, as well as the consequences of land use changes, was recently treated by Leh *et al.* (2013) and Gulickx *et al.* (2013).

According to the most commonly adopted ES classifications (see e.g. de Groot *et al.* 2002, MEA 2005, Haines-Young and Potschin 2010, Kumar 2010, UK NEA 2011, among the others), ES are defined as "the effects, influences and consequences (tangible or not tangible, quantifiable or not quantifiable) on human well-being of the internal processes and biophysical mechanisms that cyclically occur within ecosystems". Accordingly, the effects of ecosystems can be assessed and quantified in different ways (e.g. economically, ecologically, socially, politically, etc.). This definition also highlights the fundamental role of natural processes in delivering ES, which are fundamental in governing fluxes of goods and services from ecosystems to people, mainly because they can be directly measured in biophysical terms. In the same way, FES represent all outcomes (in terms of goods and services provided) of those processes and changes which occur in forest ecosystems. Deeply, the forest ecosystems potential to produce the widest range of ES over the time mainly depends on the ecological processes functioning, in which the intrinsic properties such as resilience, resistance and stability (see Box 1) in turn govern and control the fluxes of natural energy and materials. Figure 4 gives an overview of the proposed FES framework (and the correlated FES model).

Box 1: Resilience, resistance and stability of ecosystems (Holling 1973).

Resilience: The capacity of an ecosystem to return to the pre-condition state following a perturbation, including maintaining its essential characteristics, taxonomic composition, structures, ecosystem functions, and process rates.

Resistance: The capacity of the ecosystem to absorb disturbances and remain largely unchanged.

Stability: The capacity of an ecosystem to remain more or less in the same state within bounds, that is, the capacity to maintain a dynamic equilibrium in time while resisting change.

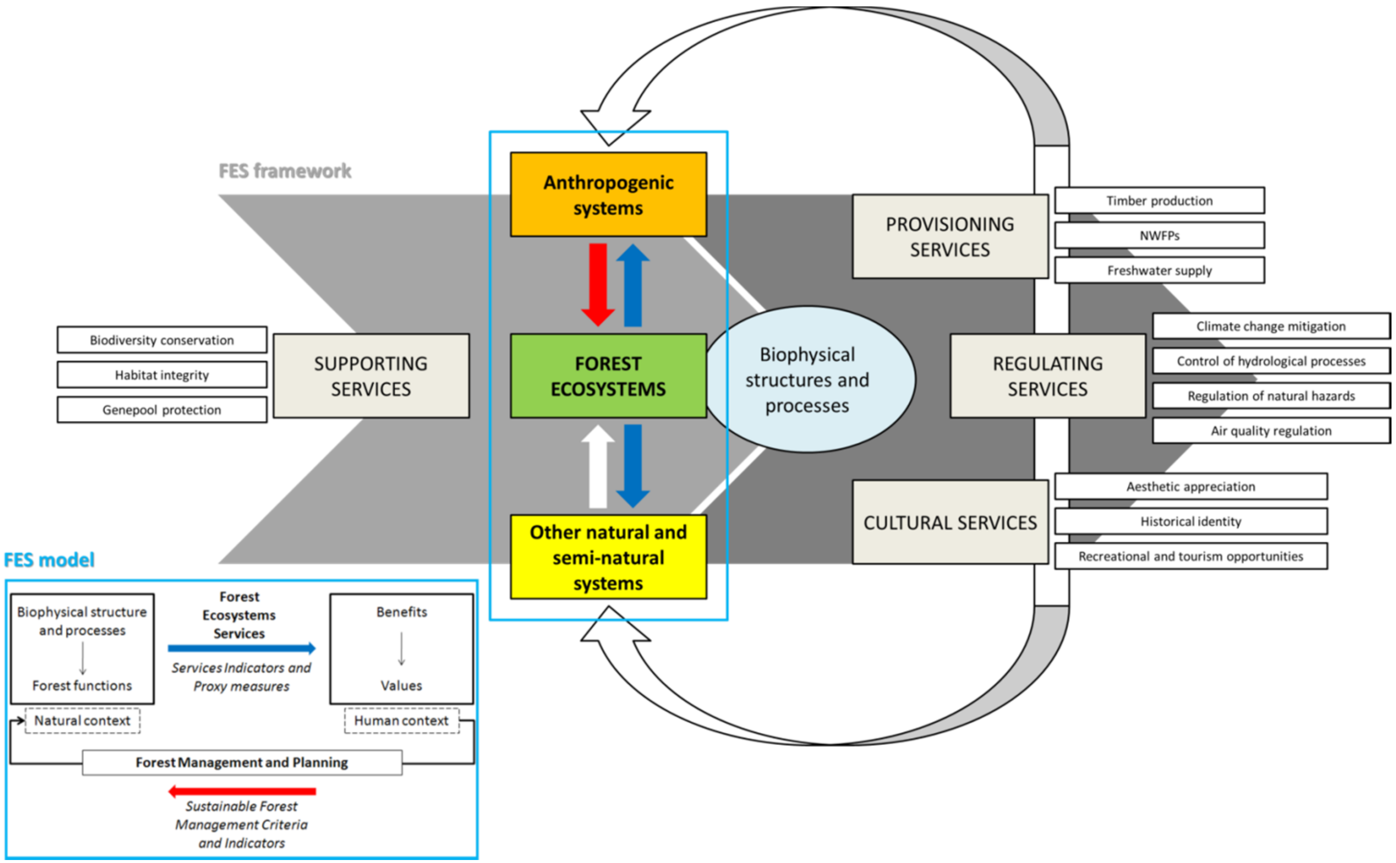

Figure 4: The FES framework (Vizzarri *et al.* 2013). The chart summarizes the most important FES (such as supporting, provisioning, regulating and cultural). Red and blue arrows explain, respectively, the *in-out* relationships (or effects) occurring between forest ecosystems and other systems (anthropogenic and other natural and semi-natural) following a holistic approach. The large arrows on background describe the direction of energy and material fluxes from supporting services (viewed here as ecosystem functions or intermediate services), which move throughout the forest biophysical structures and process, and finally originate the other final services. Adversely, semi-circular arrows represent the benefits originated by all final services on both anthropogenic and other natural and semi-natural systems. The FES model is bordered in blue. It details the relations between natural and human contexts. In particular, the FES model highlights that, while FES framework describes the fluxes of tangible and non-tangible goods and products from forests to people (as benefits), the sustainable forest management (SFM) ensures and enhances the exploitation of such benefits, as well as it describes human interventions and the anthropogenic effects on forest ecosystems. Moreover, as showed in the FES model, both FES and SFM can be assessed and measured though adopting and implementing several indicators.

2.1.1 Forests and provisioning services

Forests, other wooded lands (OWL) and trees outside forests (TOF) provide a wide range of wood and non-wood forest products (NWFPs) (FAO 2010). Research indicates that forests supply about 5,000 different commercial products (Chiras 2013), and the forestry sector contributes about 2% of GDP (FAO 1997). In addition, forested watersheds are exceptionally stable hydrological systems (FAO 2003). In comparison with other land uses, healthy forests strongly influence the quality and quantity of water yielded from watersheds (Zingari and Achouri 2007).

Forests and timber production

Timber production and the provision of wood fuel are key provisioning forest services (Harrison *et al.* 2010). They represent the economic and social utilities from forests to national, regional and local communities. For example, European roundwood production in 2007 was 728 million m^3 (33.8% of the global production) (FAOSTAT 2009). Forest products are fundamental especially for the economies of the Nordic Countries and Baltic States (EASAC 2009). The ES approach requires timber extraction to be ecologically sustainable in order to be considered as a service (Ojea *et al.* 2012a). The understanding of forest productivity and timber market provides the basis to assess and evaluate the sustainability of forest management from forest-dependent local communities, and to maintain large and valuable supplies of primary forest products (Luck *et al.* 2009, MEA 2005, Byron and Arnold 1999). Timber is only a part of the wider set of goods produced by forests, and related forest productivity theoretically refers to natural growth and yield processes (Pretzsch 2009).

Forests and non-wood forest products

Over the past two decades, NWFPs obtained from plant resources, including seeds, flowers, fruits, leaves, roots, bark, latex, resins and other non-wood plant parts, have gained much attention in conservation circles (Ticktin 2004). Hundreds millions of people world-wide currently derive a significant portion of their subsistence needs and incomes from gathered plant and animal products (Iqbal 1993; Walter 2001). As an example, Forest Europe, UNECE and FAO (2011) reported a quantity of marketed NWFPs in Italy (updated to 2005) of approximately 480,000 tons (including mushrooms, truffles, fruits, berries, edible nuts, and cork). Palahì *et al.* (2009) edited an exhaustive set of methodologies and analyses to model, value and manage Mediterranean forest ecosystems for improving NWFPs availability. In particular, Dettori *et al.* (2009) focused on the sustainability in producing and picking NWFPs in Italy.

Forests and water supply

People have settled historically in areas rich with natural resources, and today most of the world's population lives downstream of forested watersheds (Reid 2001). Societies have created strong cultural links with forests, and it is widely assumed that forests help to maintain a constant supply of good-quality water (Stolton and Dudley 2007). Moreover, the vegetation and soils of forests and wetlands have a remarkable capacity to filter out contaminants and trap sediment that would otherwise enter rivers, lakes, and streams (Postel and Thompson 2005). Forests improve the availability of water in terms of its quality, quantity and regularity (see *e.g.* Stolton and Dudley 2007). Forested watersheds generally offer high-quality water rather than alternative land uses (such as agriculture, industry, and settlements), which are likely to increase the amount of pollutants entering headwaters. In most cases, the presence of forests can substantially reduce the need for treatments (and related costs) for drinking water. Many studies suggest that in both very humid and very dry forests evaporation is likely to be greater from forests than from land covered with other types of vegetation; thus less water flows from forested catchments than, for example, from grassland or crops (Calder 2000). Constancy of flow is as important as total quantity, in terms of both maintenance of dry-season flow and absence of flooding during periods of heavy rain (Stolton and Dudley 2007).

2.1.2 Forests and regulating services

The regulating services class comprises a wide range of contributions provided by forests to control and mitigate ecological processes, external drivers and barriers, and influences and fluxes of biogeochemical materials. More generally, MEA (2005), and successively Kumar (2010), classified regulating services in the following service types: (i) air quality regulation; (ii) climate regulation; (iii) moderation of extreme events; (iv) regulation of water flows; (v) waste treatment; (vi) erosion prevention; (vii) maintenance of soil fertility and nutrient; (vii) pollination; and (viii) biological control. Of course, forests contribute to the provision of all above-mentioned ES types. Broadly, regulating services from forest ecosystems can be grouped into three main categories, such as: (i) climate change mitigation and air quality improvement (the capacity of forests to influence climate through exchanges of energy, water, carbon dioxide, and other chemical species with the atmosphere; see e.g. Bonan 2008a); (ii) hydrological processes control (the capacity of forests to control, mitigate and regulate hydrological regimes); and (iii) natural hazards regulation (the capacity of forests to mitigate extreme events that cause disasters for human population).

Climate change mitigation by forests

Forests have a unique, threefold relationship to global climate change: they are simultaneously at risk from the effects of climate change, while being part of the cause and part of the solution (Schwarze *et al.* 2002). During the last decade of the 20th Century, deforestation in the tropics and forest re-growth in the temperate zone and parts of the boreal zone remained the major factors responsible for greenhouse gasses (GHGs) emissions and removals, respectively (Nabuurs *et al.* 2007). Indeed, it is now understood that forests and human uses of forests provide important climate forcing and feedbacks (Denman *et al.* 2007), that climate change may adversely affect ecosystem functions (Fischlin *et al.* 2007), and that forests can be managed to mitigate climate change (Nabuurs *et al.* 2007). Despite the difficult to directly establish how forests can influence the large-scale climate, current generation of climate models has capability beyond hydrometeorology and incorporates ecological advances in biogeochemical and bio-geographical modeling (Bonan 2008b). Wainwright and Mulligan

(2013) provided an exhaustive description of different models used for assessing climate and climatic change at different levels. Figure 5 shows the climate forcing and feedbacks between different kinds of forest and atmosphere.

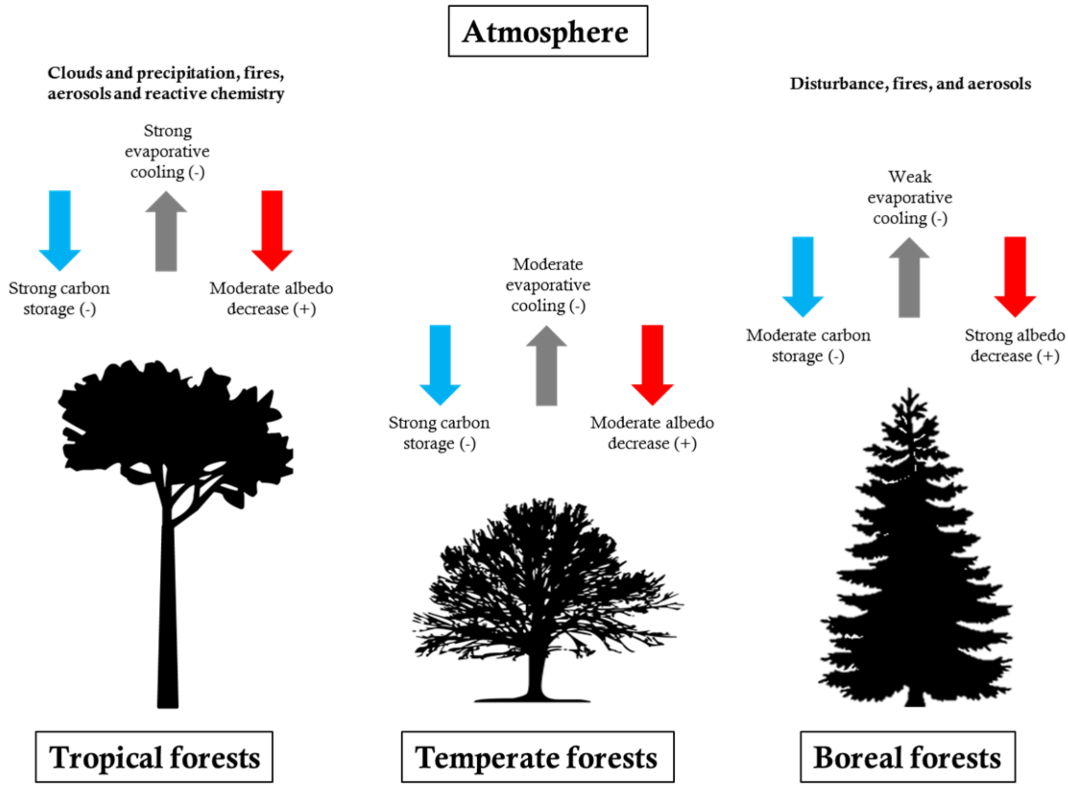

Figure 5: From left to right, climate service in (A) tropical, (B) temperate, and (B) boreal forests. Text boxes indicate key processes with uncertain climate services (Bonan 2008a, modified).

Without considering all components participating to exchanges among vegetation, atmosphere and soil, the evaluation of carbon stocked in forest stems, roots and soils can be used as a proxy measure of GHGs emissions mitigation. As an example, IPCC (2007) reported the latest estimates for the terrestrial sink for the decade 1993-2003 at 3,300 million tons CO_2 year^{-1}, ignoring emissions from land-use change (Denman *et al.* 2007), and Forest Europe, UNECE and FAO (2011) reported a value of carbon stocked in Italian forests (updated to 2010) of about 1,422 million metric tons (referring to biomass, deadwood, soil and litter).

Air quality improvement by forests

The improvement of air quality consists in the reduction of trace chemicals from the atmosphere by trees and plants, in general. Broadly, such trace chemicals are: tropospheric ozone (O_3), stratospheric O_3, nitrogen oxides (NO_X), sulfur oxides (SO_X), methane (CH_4), carbon monoxide (CO), particulates, hydroxyl radicals (OH), some heavy metals including mercury (Hg) and lead (Pb), and volatile organic hydrocarbons (VOCs). The interaction between air pollution and forests has many characteristics (Taylor *et al.* 1994), such as: (i) atmosphere-biosphere interaction; (ii) multiple pollutants; (iii) spatial distribution of pollutants; (iv) temporal horizon of pollutants; (v) rates of change in amount of each pollutant; (vi) modes of action; (vii) ecological concern; (viii) interactive effects; (ix) pollutant residence time; and (x) inadequacy of agricultural paradigm. Trees can reduce air pollutants in two ways (Yang *et al.* 2005): (i) by direct reduction from the air, and (ii) by indirect reduction by avoiding the emission of air pollutants. Directly, trees absorb gaseous pollutants through leaf stomata and also can dissolve water soluble pollutants onto moist leaf surfaces (Nowak 1994), as well as they can also intercept particulate matters in the air (Beckett *et al.* 1998). Indirectly, trees can reduce the air temperature through direct shading and evapotranspiration in the summer, thus reducing the emission of air pollutants from the process of generating energy for cooling purposes (Yang *et al.* 2005).

Therefore, reduced air temperature can lower the activity of chemical reactions, which produce secondary air pollutants in urban areas (Taha 1996; Nowak *et al.* 2000). As a source of air pollutants, trees emit biogenic volatile organic compounds (BVOCs), which can react with NO_X and form O_3 and aerosols (Benjamin and Winer 1998). In recent literature, most of the studies about air pollutants removal by forests focused on the role of urban forests in mitigating the negative effects of aerial chemical compounds on citizens' health within big cities around the world (e.g. Escobedo and Nowak 2009, Nowak *et al.* 2006, Baumgardner *et al.* 2012, etc.). In Italy, Paoletti (2009) summarized the O_3 levels along urban-to-rural gradients in three representative cities, and reviewed the state-of-knowledge of forest effects on O_3 pollution and of O_3 pollution on forest conditions in Italian cities. A review on the same topic was published also for Mediterranean forests (Paoletti 2006).

Hydrological processes control by forests

Forest ecosystems, especially in mountainous areas, are of primary importance to protect human infrastructures and buildings against avalanches, rock-fall, landslides, and mudflows, as well as general erosion phenomena (EUSTAFOR and Patterson 2011). Brauman *et al.* (2007) grouped hydrological services into four broad categories, as follows: (i) improvement of extractive water supply; (ii) improvement of in-stream water supply; (iii) water damage mitigation; (iv) provision of water-related cultural services; and (vi) water-associated supporting services. Vegetation is often the driving force in ecosystem effects on water, but all elements of an ecosystem, from microbes to mega-fauna, can and do affect hydrologic service provision (Brauman *et al.* 2007). Vegetation influences the net loading of water to the soil by intercepting precipitation, some of which is directly evaporated, attenuating radiation interception to the snowpack and forest floor, and controlling the rate of evapotranspiration (Mackay and Band 1997). Indeed, water supply (precipitation) and demand (potential evapotranspiration) are major factors affecting the long-term water balance (Budyko 1974; Milly 1994). Runoff and its components are controlled by both climatic factors and landscape properties (Horton 1933). While exposed soil surfaces on the forest floor are susceptible to splash displacement, surface runoff, and erosion (Nanko *et al.* 2006; Nanko *et al.* 2008; Nanko *et al.* 2010), the forest understory and litter layer protects soils from rainsplash erosion. The forest litter layer is also highly porous and rainfall intensity rarely exceeds infiltration rates in forested watersheds (Vose et al. 2011). Figure 6 summarizes the effects of vegetation in controlling erosion rates.

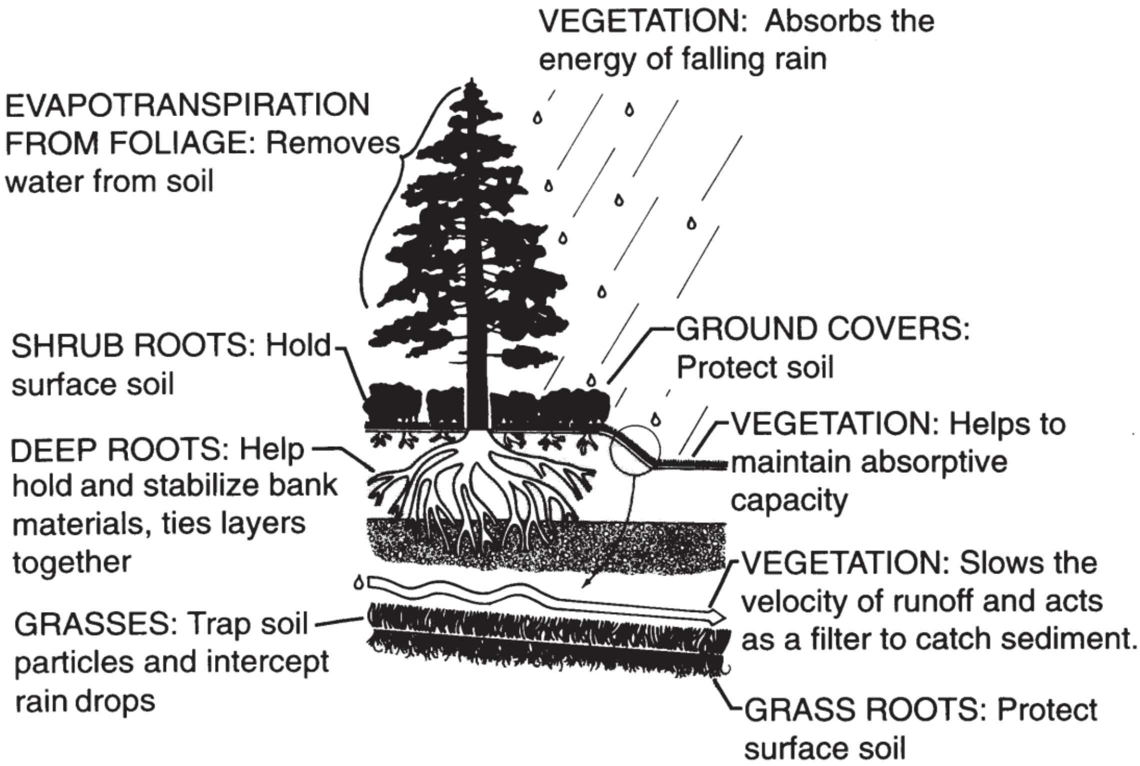

Figure 6: Summary of the effects of vegetation in minimizing the erosion (Menashe *et al.* 1993, modified).

On the other hand, forested watersheds play critical roles in regulating streamflow, despite their capacity to mitigate extreme precipitation events and reduce flooding is limited (Burt and Swank 2002; Eisenbies *et al.* 2007). Watersheds that lose forest cover exhibit increased runoff, whereas those that gain forest cover through reforestation show less runoff (Trimble *et al.* 1987). A recently published study suggests that natural forests have a larger role in flood prevention than has generally been argued of late (Bradshaw *et al.* 2007). Modeling forest hydrological processes aims to quantify (Bouten and Jansson 1995): (i) vertical soil water fluxes, as water is the main transporting agent for chemical constituents; (ii) water uptake by the forest, as transpiration is a key process in the functioning of plants; or (iii) the soil moisture condition, as it regulates a number of biological and chemical processes in the soil.

Natural hazards regulation by forests

Disturbances are those events in time that disrupt ecosystem structure, composition and/or processes by altering its physical environment and/or resources, causing destruction of plant biomass (synthesized from White and Pickett 1985; Gunderson 2000; Grime 2001; White and Jentsch 2001). Disturbances strongly influence structure, composition and functioning of forest ecosystems (Franklin *et al.* 2002) and determine the spatial and temporal patterns of forest landscapes (Forman 1995). Major natural disturbances listed by White (1979) and White and Pickett (1985) include: fire; hurricanes, windstorms and gap dynamics; ice storms, ice push, cryogenesis and freeze damage; landslides, avalanches and other earth movements, including coastal erosion and dune movement; coastal flooding; lava flows; karst processes; droughts, flash floods, rare rainstorms, fluctuating water levels, alluvial processes and salinity changes; biotic disturbances including insect attack, fungal disease, browsing and burrowing animals, invasion by plants (weeds); and disturbance caused by man. The combination of disturbance pressures can reduce biodiversity and diminish the capacity of forests to continue providing ecological goods and services of the same quantity and quality in perpetuity (Toman and Ashton 1996; Costanza *et al.* 2000). Natural disturbances are recognized as blueprints for close-to-nature management, assuming that the ecosystem and its components (e.g. endangered species) are resilient to disruptions that closely mimic natural dynamics (e.g. Palik *et al.* 2002; Bouchard *et al.* 2008). The capacity of environmental but also societal systems to cope with disturbances while maintaining their main functions, structures, identities and feedbacks is described as a system's *resilience* (Walker *et al.* 2006). Indeed, the resilience of ecosystems (see Box 1) may be an essential factor underlying the sustained production of natural resources and ES in complex systems faced with uncertainty and surprise (Gunderson and Holling 2002).

Sustaining desirable states of an ecosystem in the face of compounded perturbations requires that functional groups of species remain available for renewal and reorganization (Lundberg and Moberg 2003). Research and experience have shown that forest ecosystems play an important role in reducing the vulnerability of communities to disasters, both in terms of reducing their physical exposure to natural hazards and providing them with the livelihood resources to withstand and recover from crises (Hammill *et al.* 2005). Decreasing in forest ecosystems resilience (as 'degradation') can aggravate the human consequences of natural disasters (MEA 2005). Whereas some connections of ecosystem change to disasters are evident, there are little quantitative information with which to measure the disaster risks associated with ecosystem change (Carpenter and Folke 2006). Thus, fundamental research is still needed to improve planning to avoid or mitigate future natural disasters.

2.1.3 Forests and biodiversity conservation

The United Nations Convention on Biological Diversity (CBD) defined biodiversity as "the variability among living organisms from all sources, including, inter-alia, terrestrial, marine, and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems" (United Nations 1992; p. 3). This definition emphasizes the variability at three levels (Mace *et al.* 2012): (i) within species (thus including genetic- and population-level measures); (ii) between species (all measures of species-level variation); (iii) within ecosystems (thus including measures at landscape or regional levels, such as major vegetation types or biomes). Species diversity, vertical structural diversity, and horizontal structural diversity together comprise forest structural diversity (Pommerening 2002; Varga *et al.* 2005) which, in turn, constitutes one of three primary components of biological diversity (Noss 1990).

Biological diversity occurs at all spatial scales, from local through regional to global (Probst and Crow 1991). ES are measured irrespective of the way that biodiversity contributes to them and the conservation of species is considered alongside and potentially in opposition to other benefits, such as flood regulation, carbon sequestration or agricultural productivity on the same parcel of land (Eigenbrod *et al.* 2009, Kumar 2010). More than its intrinsic value, the roles of biodiversity for the exploitation of ES can be summarized by the following headings (MEA 2005): (i) supporting roles include the underpinning of ecosystems through structural, compositional, and functional diversity; (ii) regulatory roles through the influence of biodiversity on the production, stability, and resilience of ecosystems; (iii) cultural roles from the nonmaterial benefits people derive from the aesthetic, spiritual, and recreational elements of biodiversity; and (iv) provisioning roles from the direct and indirect supply of food, fresh water, fiber, and so on. Moritz (2002) stated that the overarching aim of conservation biology is to protect biological diversity and the processes that sustain it in the face of the perturbations caused by human activities.

According to Noss (1999), some important objectives for conservation of forest biodiversity should include: (i) representing all kinds of communities or ecosystems, across their natural range of variation; (ii) maintaining or restoring viable populations of all native species in natural patterns of abundance and distribution; (iii) sustaining key geomorphological, hydrological, ecological, biological, and evolutionary processes within normal ranges of variation, while being adapted to a changing environment; and (iv) encouraging human uses that are compatible with the maintenance of ecological integrity, and discourage those that are not (Noss and WWF 1995). While there are multiple functions that regulate services from ecosystems, few studies have investigated the role of biodiversity for multiple ecosystem functions jointly (Hector and Bagchi 2007; Gamfeldt *et al.* 2008; Zavaleta *et al.* 2010; Maestre *et al.* 2012), and none of them has focused on services *per se* (Gamfeldt *et al.* 2013). In a recent comprehensive review, a majority of the included ES was related to biodiversity in the direction expected from predictions (Cardinale *et al.* 2012). However, for many of the studied services, the evidence for beneficial effects of biodiversity was mixed, or there were not enough data for a thorough evaluation (Cardinale *et al.* 2012). Thus, Díaz *et al.* 2006 proposed four suggestions for filling the gaps in biodiversity knowledge, such as: (i) deeper understanding of the links between biodiversity and the other services, especially in the species-richest ecosystems; (ii) better model building to anticipate or avoid undesiderable ecological surprises; (iii) a systematic screening reinforcement for functional traits of organisms likely to have ecosystem-level consequences; and (iv) mimicking real biotic situations in experimental designs as a result of common land use practices.

2.1.4 Forests' amenities, cultural values and recreational activities

Cultural services have been identified as "cultural diversity, spiritual and religious values, knowledge systems, educational values, inspiration, aesthetic values, social relations, sense of place, cultural heritage values, recreation and ecotourism" (MEA 2005). Daniel *et al.* (2012) distinguished 4 research areas within the cultural services framework, such as: (i) landscape aesthetics; (ii) cultural heritage; (iii) recreation and tourism; and (iv) spiritual and religious significance. Aesthetic values of forests relate to preferences people have for beholding and experiencing forests (Gobster 1999). Aesthetic preferences might be directed toward particular forest features such as large trees or waterfalls; spaces that have special mining because of their location, history, or symbolism; or landscapes and ecosystems characterized by their particular qualities, processes or functions (Gobster and Chenoweth 1989). Cultural ES create strong ties between humans and their natural surroundings and play a crucial role in "feeling at home" in a landscape (Schaich *et al.* 2010). Moreover, cultural services represent one of the strongest incentives for people in developed countries to become involved in environmental conservation (Philips 1998).

Tourism is defined as the sum of the processes, activities, and outcomes arising from the relationships and the interactions among tourists, tourism suppliers, host governments, host communities, and surrounding environments that are involved in attracting, transporting, hosting and the management of tourists and other visitors (Weaver 2010). The essence of a natural experience is a combination of the sights (such as of natural vegetation, wildlife and wilderness landscapes), the sounds (such as bird song, insect, and amphibian soundscapes, the calling of mammals), and the smells (of wildflowers, seashores), as well as the state of mind it induces (Newsome and Moore 2012). Protected natural areas are now among the most sought after tourist attractions (Butler and Boyd 2000), because their protected status ensures their naturalness.

2.2 Ecological and economic foundations in assessing forest ecosystem services

2.2.1 From biophysical components to services provision

Theoretically, biodiversity reflects the hierarchy of increasing levels of organization and complexity in ecological systems; namely at the level of genes, individuals, population, species, communities, ecosystems and biomes (Chapin III *et al.* 2011). Communities of organisms interact with the abiotic environment, thus comprising and characterizing ecosystems. In turn, ecosystems are varied both in size and complexity, and may be nested one within another. According to Tansley (1935) and Odum (1969), the ecosystem model implies comprehensive understanding of the interactions responsible for distinctive ecosystem types, but unfortunately this knowledge is rarely available (Elmqvist *et al.* 2010). As a result, the use of the term ecosystem, in the case of e.g. forests, is more conceptual than based on any distinct spatial configuration of interactions. The population dynamics of species create temporal and spatial heterogeneity, while gradients in abiotic variables add to the latter (Whittaker 1975), often over orders of magnitude (Ettama and Wardle 2002). Ecosystem processes result from the life processes of multi-species assemblages of organisms and their interactions with the abiotic environment, as well as the abiotic environment itself (Elmqvist *et al.* 2010). These processes ultimately generate services when they provide benefits to humans (see Table 4).

Table 4: Some examples of biological and physical processes and interactions that comprise ecosystem functions important for ES (Virginia and Wall 2001).

Ecosystem function	Process
Primary production	Photosynthesis Plant nutrient uptake
Decomposition	Microbial respiration Soil and sediment food web dynamics
Nitrogen cycling	Nitrification Denitrification Nitrogen fixation
Hydrologic cycle	Plant transpiration Root activity
Soil formation	Mineral weathering Soil bioturbation Vegetation succession
Biological control	Predator-prey interactions

The idea of a ‘service cascade’ (Figure 7) can be used to summarize much of the logic that underlies the contemporary ecosystem service paradigm and key elements of the debate that has developed around it (Haines-Young and Potschin 2010). The model attempts to capture the prevailing view that there is something of a ‘production chain’ linking ecological structures and processes on the one hand and elements of human well-being on the other, and that there are potentially a series of intermediate stages between them (Haines-Young and Potschin 2009).

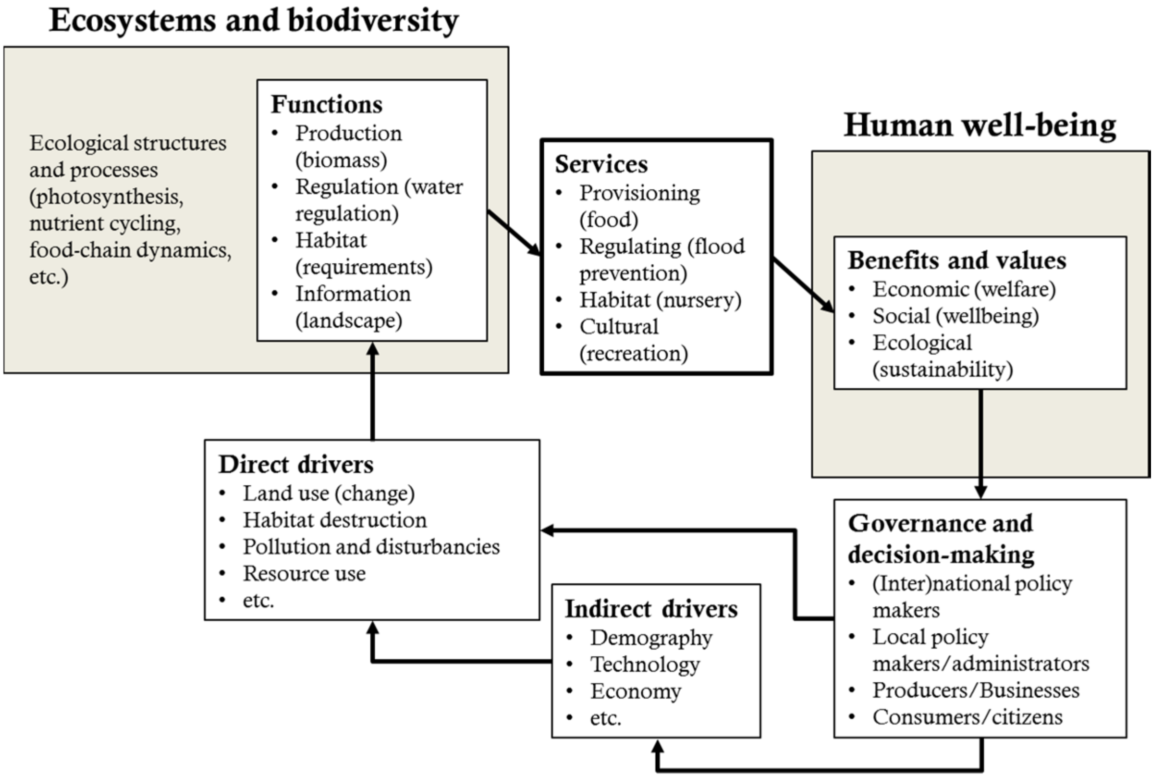

Figure 7: The 'service cascade' representation (Haines-Young and Potschin 2010; de Groot *et al.* 2010a; Kandziora *et al.* 2013, modified).

The 'service cascade' model roughly reproduces the energy flow going down from the ecosystem functions (originated by the ecosystem asset) which originate goods and services, and finally produce valuable human benefits. In conjunction with the economic incomes they produce, ES have to be balanced with both biotic and abiotic drivers that can directly influence the ecosystem functioning.

A large number of studies demonstrated that biodiversity increases and stabilizes productivity (Weigelt *et al.* 2008; Tilman and Lehman 2006), and increases soil carbon sequestration (Steinbeiss *et al.* 2008), nutrient retention (Scherer-Lorenzen *et al.* 2003), and stability of multiple functions (Hooper *et al.* 2005). Complementary resource use rather than selection of high performing species by chance (sampling effect) was identified as the main driver of these positive diversity effects (Hooper *et al.* 2005). Concerning forest ecosystems, Nadrowski and Scherer-Lorenzen (2010) confirmed that species-rich forests generally show higher productivity than species-poor forests (see also Thompson *et al.* 2009; Caspersen and Pacala 2001). Additionally, Gamfeldt *et al.* (2013) demonstrated that, in temperate and boreal forests, the relationships between tree species richness and multiple ES is positive, and that all services attain higher levels with more tree than with one species. As also pointed out by Naeem (2006) and Cardinale *et al.* (2012), understanding and managing the complexity of biological diversity and the ecosystem functioning (in terms of stability of key components and processes) are needful to better realize the full potential of several economically, ecologically and culturally valuable ES (see also Gamfeldt *et al.* 2013).

2.2.2 Trade-offs and ecosystem services evaluation

In economics, 'value' is always associated with trade-offs – that is, something only has (economic) value if we are willing to give up something in order to get or enjoy it (de Groot *et al.* 2010b). Used as the most common metric in economics, the monetary evaluation often fails to incorporate several types of value that are critical to understanding the relationship between society and nature (e.g. Norgaard and Bode 1998; Wilson and Howarth 2002; MEA 2005; Christie *et al.* 2006). In addition to the monetary evaluation, livelihoods assessment, capabilities approaches, and vulnerability approach can emphasize the opportunities to people to make choices (e.g. Sen 1993). Since there are multiple theories of value, valuation exercises should ideally: (i) acknowledge the existence of alternative, often conflicting, valuation paradigms; and (ii) be explicit about the valuation paradigm that is being used and its assumption. Two approaches can be used for valuation, such as (Pascual *et al.* 2010): (i) biophysical methods, which use a 'cost of production' perspective that derives values from measurements of physical costs (e.g. in terms of labor, surface requirements, energy or material inputs, etc.) of producing a given good or service; and (ii) preference-based methods, which rely on models of human behavior and rest on the assumption that values arise from the subjective preferences of individuals (see Figure 8).

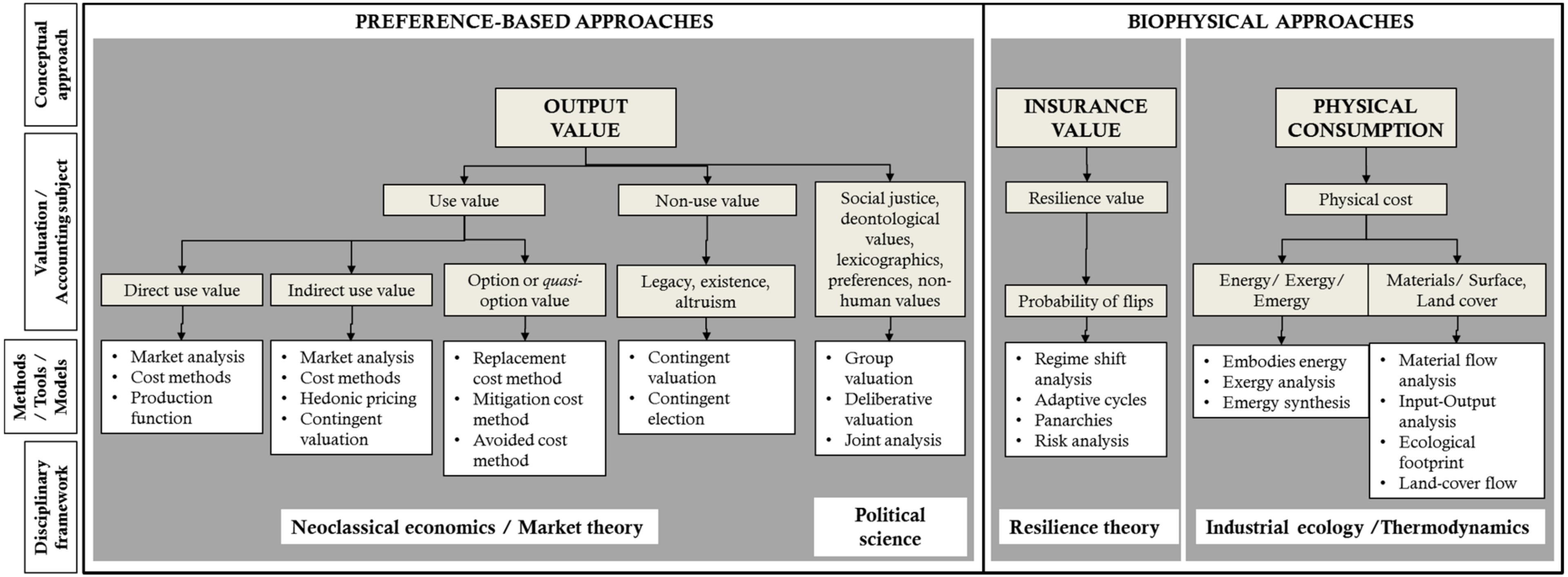

Figure 8: Approaches for the estimation of nature's values. Source: Gómez-Baggethun and de Groot (2010), modified.

In making decisions at any level (private, corporate or government), decision makers are faced with the dilemma of how to balance (weighting) ecological, socio-cultural and economic values of ecosystems (e.g. Rodríguez *et al.* 2006; Martín-López *et al.* 2014). Preferably, the importance of each of these value-components should be weighted on its own (qualitative and quantitative) dimension, through e.g. the Multi-Criteria Decision Analysis (MCDA; e.g. Schwenk *et al.* 2012), the Cost-Benefit Analysis (CBA, Wegner and Pascual 2011), and the Cost-Effectiveness Analysis (CEA, Birch *et al.* 2010). These approaches are commonly used for the trade-offs analysis, namely balancing competitive services. A trade-off occurs when the extraction of a service is negative for the provision of other services. For example, timber extraction from a forest will affect vegetation structure and composition, aesthetics and water qualities, which will preclude the continuous provision of other services, such as carbon sequestration or recreation (Pascual *et al.* 2010).

In the case of forest ecosystems, Duncker *et al.* (2012) gave an exhaustive example of the complex synergy and tradeoff patterns between production and the other ES, and within the other ES. Similarly, Cademus *et al.* (2014) provided a repeatable and simplified approach to identify specific areas where synergies occur among different ecosystems services provided by a forest stand dominated by a single tree species (i.e. pine plantations). More broadly, Ninan and Inoue (2013) estimated the value of FES across forest sites, countries, and regions (see Table 5).

Table 5: Summary of the annual value of FES (Ninan and Inoue 2013, modified). *Purchasing Power Parity series compiled by the World Bank.

FES	Range of values [2010 PPP* US$ ha^{-1}]	Mean	Median
Watershed protection/hydrological services	5–1160	248	174
Soil conservation	3–910	210	43
Carbon sequestration/gas regulation	4–3400	733	203
Recreation	2–279	41	16
Waste treatment/environmental purification	8–755	261	20
Nutrient cycling	56–228	142	142
Pollination services	205–434	320	320
Other services (pharmaceutical, biodiversity, primary productivity, etc.)	1–789	189	35
Total value	8–4080	753	441

2.3 Case study 1: A downscaled review on forest ecosystem services in Italy[3]

2.3.1 The context

Globally, forests cover more than 3.8 billion ha (Schmitt *et al.* 2009) and provide ecosystem goods and services accounting for more than 9,000 \$ ha^{-1} $year^{-1}$ (de Groot *et al.* 2012). At European level, forests and other wooded land occupy 177 million ha (42% of the EU-27 land area), of which 89 million ha are used to obtain wood and other products for the market (ForestEurope, UNECE and FAO 2011), and in Italy forests cover more than 10 million ha (about 30% of the national land area; Gasparini *et al.* 2010), of which 27.5% are protected (Gasparini and Tabacchi 2011).

The debate around ES has rapidly increased over the last two decades. A large part of researches concerning FES has been, however, generally focused on policy measures and decision-making processes-related issues, and in particular on (i) improving the availability of FES at different scales, from landscape to global level (e.g. Deal *et al.* 2012; Liu *et al.* 2013) and (ii) preserving biodiversity and habitats or enhance the ecosystem functionality (e.g. Prato 2009; DeClerck *et al.* 2010; Freudenberger *et al.* 2012; Onaindia *et al.* 2013). In most cases, forest management is ancillary to other issues (e.g. Gren and Isacs 2009; Holl and Aide 2011; Dymond *et al.* 2012; Ojea *et al.* 2012a). Studies focusing on specific forest ecosystem processes (e.g. Maes *et al.* 2013a; Hanson *et al.* 2013; Willaarts *et al.* 2012), as well as on the effects of land use change on FES provision (Martínez *et al.* 2009, Fu *et al.* 2013, Band *et al.* 2012, Leh *et al.* 2013 and Gulickx *et al.* 2013) are still poorly available.

There is no exception in Italy. In this case, despite forest ecosystem functions, goods and services have been mainly linked overtime to the concepts of multi-functionality, naturalness and biodiversity conservation (Fabbio *et al.* 2003), related research contributions have been scarce and in many cases referred to the assessment of the whole set of FES (see e.g. Busch *et al.* 2012, Paletto and Chincarini 2012, Santopuoli *et al.* 2012, and Palliggiano *et al.* 2012, etc.) or to their economic evaluation (see e.g. Notaro and Paletto 2011; Horton *et al.* 2003; Gatto *et al.* 2009; Pettenella *et al.* 2012, etc.).

2.3.2 Objectives and methodology

A step-by-step literature review is carried out to unravel the state of knowledge about FES by downscaling from global to Italian level, and by analyzing the aims and contents of national studies in comparison with those available at a broader scale. The literature review is based on a *by-keywords* basic search using SCOPUS (www.scopus.com) as unique search tool. This choice was pursuit to avoid confusion on interpreting the results by using more than one search tool. The review on EU-funded research projects is based on a free text search using the Community Research and Development Information Service (CORDIS) of the European Commission (cordis.europa.eu). For both of the reviews, the reference time period was fixed from 2000 to 2012. The main evaluation parameters are: (i) the number of publications per

[3] Vizzarri *et al.* (2014, *submitted*).

year;; (ii) the number of citations per publication; and (iii) the analysis of the main contents per publication. The review is structured into 4 hierarchical levels, named here Review Sections (RS). Each RS is composed by different Search Steps (SS), which are singularly described by the keywords used in the search strength. From upper to lower level: (i) RS A refers to the overview of the scientific contributions concerning the ecosystem or environmental services, and their linkages with forests, mainly at global level; (ii) RS B and C are deeper oriented to analyze the different service classes (both generally for all ecosystems and specifically for forests); (iii) RS D concerns the publications about ES (and forests) at national level, in Italy; and (iv) RS E refers to the number of available projects (concluded or currently at work) strictly linked to the FES topic. Table 6 reports the main methodological characteristics of the literature review.

Table 6: Details on the procedure adopted for the literature review.

Review Section (RS)	Search engine	Search Step (SS)	Acronym (used in legends)	Search strength	Evaluation parameters (expected results)
A	SCOPUS	1	ES	"ecosystem services" OR "environmental services"	number of publications per year (n pub year^{-1}); number of citations per publication (n cit pub^{-1})
		2	ES-F	"ecosystem services" OR "environmental services" AND "forests"	
		3	FES	"forest ecosystem services" OR "forest environmental services"	
B		1	PROV-E	"provisioning services" AND "ecosystems"	
		2	REG-E	"regulating services" AND "ecosystems"	
		3	BIO-E	"biodiversity services" OR "habitat services" OR "supporting services" AND "ecosystems"	
		4	CULT-E	"cultural services" OR "aesthetic services" OR "amenity services" OR "tourism services" OR "recreational services" AND "ecosystems"	
C		1	PROV-FE	"provisioning services" AND "forests" OR "forest ecosystems"	
		2	REG-FE	"regulating services" AND "forests" OR "forest ecosystems"	
		3	BIO-FE	"biodiversity services" OR "habitat services" OR "supporting services" AND "forests" OR "forest ecosystems"	
		4	CULT-FE	"cultural services" OR "aesthetic services" OR "amenity services" OR "tourism services" OR "recreational services" AND "forests" OR "forest ecosystems"	
D		1	ES-IT	"ecosystem services" AND "Italy"	analysis of the main contents (n pub TA^{-1});
		2	FES-IT	"forest ecosystem services" AND	

Review Section (RS)	Search engine	Search Step (SS)	Acronym (used in legends)	Search strength	Evaluation parameters (expected results)
				"Italy"	number of citations per publication (n cit pub^{-1})
		3	ES-F-IT	"ecosystem services" AND "forests" AND "Italy"	
E	CORDIS	1	FES	"forest ecosystem services" (AND "Italy")	Number of projects; main contents and objectives

The publications founded in RS D have been then grouped into specific thematic areas (TAs), through verifying the consistency of their contents with the theoretical concepts and aims behind each TA. We selected the following TAs: (i) ES assessment (approaches, techniques and methods); (ii) ES role in the policy context; (iii) ES in urban (and semi-natural) areas; (iv) ES and local communities; (v) The economics of ES; and (vi) ES and Land Use, Cover and Change (LUCC).

Such TAs have been chosen for their significance in the ES-related research. Following this approach, publications have been analyzed by their number for a specific TA in agreement with their contents.

2.3.3 Results

Through the review process we found more than 9,000 records, of which about 73% refers to articles, 11% to reviews and 15% to other document types. Table 7 summarizes the main outcomes of our review. No results were found for SS D.2.

Table 7: Summary of review results in terms of number of records found per document type. Relative percentages are reported in brackets.

RS	SS	Total results	Papers	Reviews	Other document types
A	1	7010	5020 (0.72)	823 (0.12)	1167 (0.17)
	2	1618	1280 (0.79)	146 (0.09)	192 (0.12)
	3	142	119 (0.84)	10 (0.07)	13 (0.09)
B	1	68	51 (0.75)	6 (0.09)	11 (0.16)
	2	56	48 (0.86)	6 (0.11)	2 (0.04)
	3	64	46 (0.72)	10 (0.16)	8 (0.13)
	4	102	77 (0.75)	15 (0.15)	10 (0.10)
C	1	16	12 (0.75)	1 (0.06)	3 (0.19)
	2	13	11 (0.85)	2 (0.15)	0 (-)
	3	12	9 (0.75)	1 (0.08)	2 (0.17)
	4	37	33 (0.89)	2 (0.05)	2 (0.05)
D	1	25	20 (0.80)	1 (0.04)	4 (0.16)
	2	0	0 (-)	0 (-)	0 (-)
	3	8	7 (0.88)	0 (-)	1 (0.13)
E	Detailed results in Appendix 1				

From the review results, we selected 350 papers as most relevant in understanding the ES research topic, and according to their relative citations, publishing dates and keywords. We analyzed them in terms of their contents, results, conclusions, and relevance within the ES topic. Additionally, in RS D we reviewed 34 publications, previously grouped into different TAs.

Detailed review results per RS are hereinafter reported.

Section A

Figure 9 reports the number of publications from 2000 to 2012 for SS A.1-3.

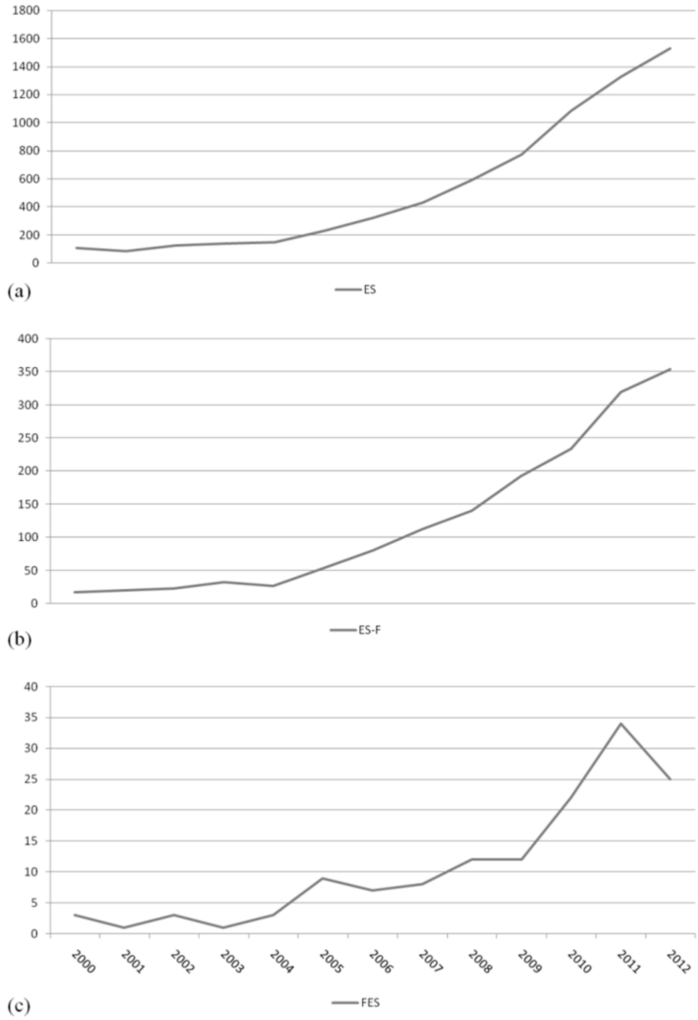

Figure 9: Trends of the n pub year^{-1} as resulted in RS A and for SS A.1 (a), SS A.2 (b), and SS A.3 (c).

Considering the SS A.1, the number of publications increased of about 1,300 units after the MA in 2005 (Figure 9(a)). SCOPUS registered a total number of 1,531 publications in 2012. Considering the SS A.2, the number of publications increased of about 300 units after the MA in 2005 (Figure 9(b)). SCOPUS reported a total number of 354 publications in 2012. Considering the SS A.3, the number of publications increased of 25 units after MEA in 2005 and until 2011 (Figure 9(c)). After a peak in that year (34 publications), SCOPUS reported a decrease in the total number of publications, down to 25 in 2012. These results demonstrate the global interest by research community on ES-related topics, especially after the release of MEA in 2005. Nevertheless, the number of publications related to SS A.2 and A.3 is, respectively 23% and 2% of the total number of publications obtained in SS A.1.

Table 8 reports the results concerning the number of citations per SS A.1-3 and per best-cited publication, both for the 2000-2012 period and after 2012.

Table 8: Total number of citations for the best-cited reference as resulted by RS A. (1) The total number of citations is referred to the top 10 cited publications for each SS.

Review Section (RS)	Search Step (SS)	Total number of citations (1)		Best-cited reference	Number of citations per best-cited reference	
		2000-2012	>2012		2000-2012	>2012
A	1	6536	915	Hooper *et al.* 2005	1387	173
	2	1936	394	Allen *et al.* 2010	237	137
	3	412	52	Grieg-Gran *et al.* 2005	88	10

Sections B and C

Figure 10 reports the number of publications from 2000 to 2012 for SS B.1-4 and for SS C.1-4.

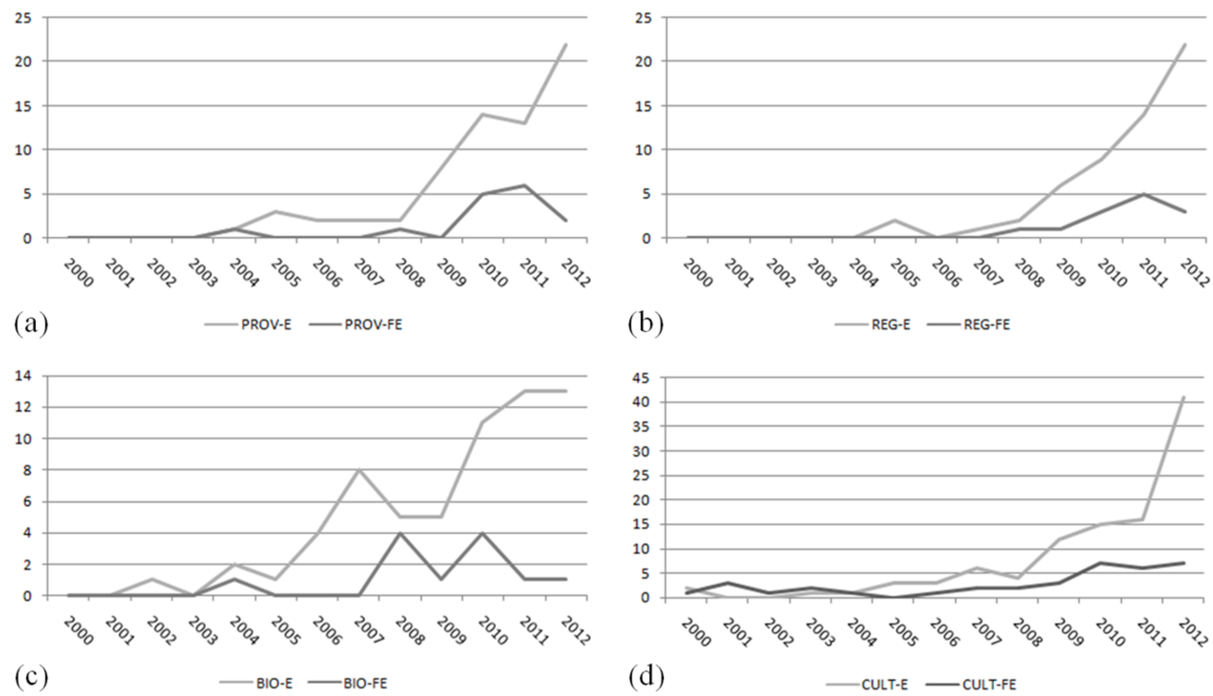

Figure 10: Trends of the n pub year^{-1} as resulted in RS B and C and for: (a) SS B.1 and C.1; (b) SS B.2 and C.2; (c) SS B.3 and C.3; and (d) SS B.4 and C.4.

Considering SS B.1 and B.2, results show that the number of publications rapidly increased after 2008 (PROV-E and REG-E, Figure 10(a, b)). In both of these cases, they passed from 2 in 2008 to 22 in 2012. A similar trend regards SS C.1 and C.2 (PROV-FE and REG-FE, Figure 10(a, b)). Even in these two cases, the number of publications increased from 1 in 2008 to 6 in 2011 (for SS C.1) and from 1 in 2008 to 5 in 2011 (for SS C.2). After 2011, there was a decrease in the number of publications, up to 2 (for SS C.1) and 3 (for SS C.2). Considering SS B.3 and B.4, results show that the number of publications generally increased from 2005 to 2012 (with different trends during this period). In particular, the number of publications for B.3 passed from 1 in 2005 to 13 in 2012, and for B.4 it passed from 3 in 2005 to 41 in 2012 (BIO-E and CULT-E, Figure 10(c, d)). A different trend describes the results of SS C.3 and C.4 (BIO-FE and CULT-FE, Figure 10(c, d)). In the first case, the number of publications fluctuated from 0 to 4 in the 2005-2008 period (only one publication was released before, in 2004), and then from 4 to 1 in the 2010-2012 period. In the second case, the number of publications generally increased from 0 in 2005 to 7 in 2012.

Table 9 reports the results concerning the number of citations for SS B.1-4 and C.1-4, and per best-cited publication, both for the 2000-2012 period and after 2012.

Table 9: Total number of citations for the best-cited reference as resulted by RS B and C. ([1]) The total number of citations is referred to the top 10 cited publications for each SS.

Review Section (RS)	Search Step (SS)	Total number of citations ([1])		Best-cited reference	Number of citations per best-cited reference	
		2000-2012	>2012		2000-2012	>2012
B	1	316	138	Zhang *et al.* 2007	73	34
	2	230	109	Zhang *et al.* 2007	73	34
	3	408	131	Rodríguez *et al.* 2006	80	32
	4	508	175	Wallace 2007	132	33
C	1	56	21	Shulz *et al.* 2010	20	7
	2	51	21	Harrison *et al.* 2010	13	7
	3	63	18	Feld *et al.* 2009	16	11
	4	166	32	Elands and Wiersum 2001	35	4

Section D

Figure 11 shows the sharing of publications per TA as resulted by SS D.1 and SS D.3. No results have been found for SS D.2.

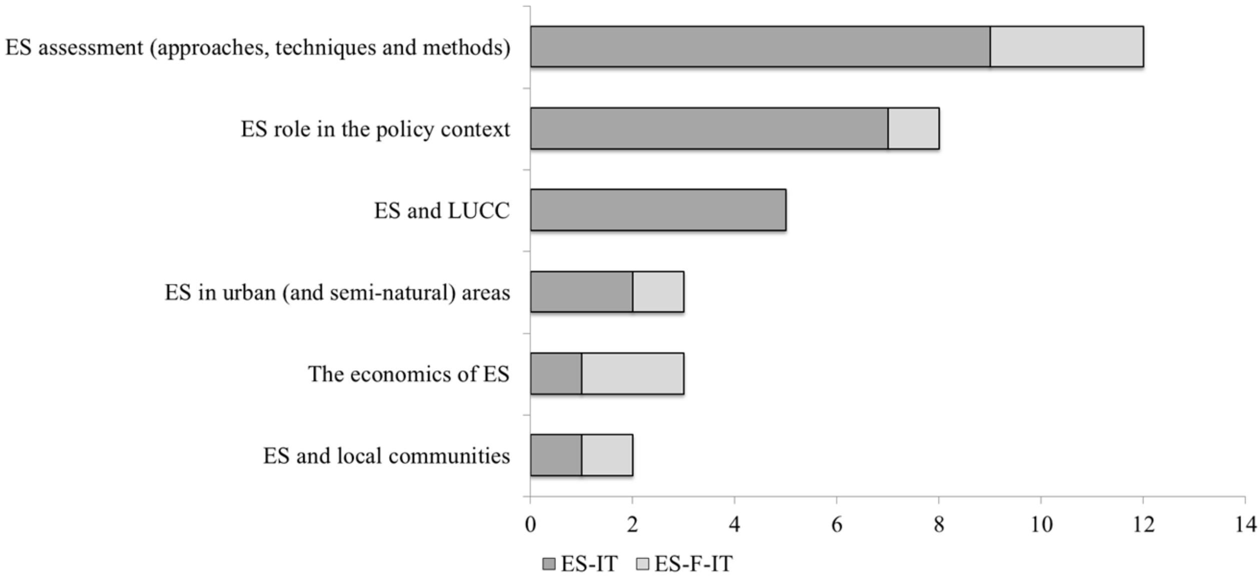

Figure 11: Number of publications per TA as resulted in RS D.

Considering RS D, the total number of publications in the 2000-2012 period was 25 (for SS D.1) and 8 (for SS D.3) (ES-IT and ES-F-IT, respectively, Figure 3). By analyzing the main contents, the congruence of the reviewed publications to the identified TAs can be summarized with the following headings: (i) most of publications concern "ES assessment TA" (9 for SS D.1 and 3 for SS D.3); (ii) very few publications regard 'The economics of ES' and 'ES and local communities' TAs (1 for SS D.1 and, respectively, 2 and 1 for SS D.3); (iii) no publications for SS D.3 are correlated to the 'ES and LUCC' TA.

Table 10 reports the results concerning the number of citations per RS D.1-3 and per best-cited publication, both for the 2000-2012 period and after 2012.

Table 10: Total number of citations for the best-cited reference as resulted by RS D. ([1]) The total number of citations is referred to the top 10 cited publications for each SS.

<table>
<tr><td rowspan="2">Review Section (RS)</td><td rowspan="2">Search Step (SS)</td><td colspan="2">Total number of citations ([1])</td><td rowspan="2">Best-cited reference</td><td colspan="2">Number of citations per best-cited reference</td></tr>
<tr><td>2000-2012</td><td>>2012</td><td>2000-2012</td><td>>2012</td></tr>
<tr><td rowspan="3">D</td><td>1</td><td>68</td><td>32</td><td>Horton et al. 2003</td><td>20</td><td>1</td></tr>
<tr><td>2</td><td colspan="5">No results were found.</td></tr>
<tr><td>3</td><td>25</td><td>10</td><td>Horton et al. 2003</td><td>20</td><td>1</td></tr>
</table>

Section E

Considering RS E, the results can be summarized as follows: (i) since 2000, the total number of the EU-funded projects focusing on ES and, in particular, on forest resources, is 68, of which 29% will end in the *post*-2012 period; (ii) Italy is included in 24 of the project consortia, and is coordinator for three of them; (iii) for 25% of the projects the main aims and research activities are consistent with the FES topic, totally or partially (10.3% and 14.7%, respectively), while they are not specifically oriented to the FES topic (approximately 29%). For detailed results, the reader is referred to Appendix 1.

2.3.4 Discussion and conclusions

Generally, the literature review and the methodology chosen (about its structuring and the keywords used) tried to be as inclusive as possible. Through a downscaled approach, the review focused on the state-of-the art and trends about ES and FES related research, from global to Italian level, and from the whole set to specific services. The review outcomes are discussed according to the main issues (or lacks of information).

State of knowledge about ES and FES research

Results from RS A demonstrate that there is a global lack of knowledge in assessing, quantifying or evaluating FES, as well as in treating forests as separate ecosystems. Despite the increasing global awareness among scientists on the ES topic since the MEA release in 2005 (as also pointed out by Daily and Matson 2008, and Nieto-Romero *et al.* 2014), the number of publications on FES has been relatively stable till now (see Figure 9). Moreover, FES-related publications were 10 times less than those obtained for ES and ES-F (see Figure 9). This explains that the role of forests in the ES framework is not completely treated or widespread or even considered of primary importance.

Detailing research by FES type

By downscaling the review and analyzing ES separately (as for RS B and C), the best-cited publications are not always consistent with the service type as expected, excepting than for biodiversity conservation (BIO-E and BIO-FE). In the case of RS B, some examples are (i) Zhang *et al.* (2007) for provisioning (PROV-E) and regulating services (REG-E), who discussed the services from agriculture; and (ii) Rodríguez *et al.* (2006), who assessed the trade-offs of ES according to different scenarios (see Table 9). When focusing on forests (RS C), the most interesting examples in this sense are (i) Shultz *et al.* (2010) for provisioning services from forests (PROV-FE), who mainly focused on forest cover changes; and (ii) Elands and Wiersum (2001) for cultural services from forests (CULT-FE), who discussed the perceptions of the potential role of forestry in rural development (see Table 9). Even considering RS B and C, cultural services (CULT-E and CULT-FE) are treated into a relatively high number of publications (about the double in comparison with the other service categories) (see Figure 10). It is partially explained by the fact that cultural services have gained more attention over the last years (see e.g. Chiesura and de Groot 2003; Martín-López *et al.* 2012; Milcu *et al.* 2013). This inconsistency between the search strength and obtained results may depend on the level of detail of the used keywords, as well as on the search engine itself. Moreover, the unstructured results by service type may be originated by the tendency to treat different services as integrated parts of a "whole group". As a consequence, scientists writing a paper on the ES topic hopefully need a broader perspective, which is mainly provided by the most-cited references reported above. This kind of approach (in agreement with our results) explains that ecosystems globally have a trans-disciplinary role, which ranges from the socio-economic, to the biophysical, and to the policy and planning contexts (see e.g. Cowling *et al.* 2008).

The important role of biodiversity in FES research

Globally, biodiversity is a key term in the ES and ES-F contexts. This consideration is justified by the largest amount of citations for Hooper *et al.* (2005) in RS A-ES (see Table 8) and for Feld *et al.* (2009) in RS C-ES-F (see Table 9). Regarding forest resources, Thompson *et al.* (2009) reported that 76% of 21 reviewed studies showed a direct relationship between increased biodiversity (measured as tree and understory species richness) and increased primary productivity. In the same way, Balvanera *et al.* (2006) and Thompson *et al.* (2009) confirmed that plant diversity enhances belowground plant and microbial biomass and decomposer activity and diversity, resulting in greater diversity of primary consumers and a lower number of invasive species relative to systems with low levels of productivity. Gamfeldt *et al.* (2013) found consistent positive relationships between tree species richness (contrasting plots with five and one tree species) and multiple ES, thus confirming that the conservation of forest stand diversity is needed to safeguard a future potential of high levels of multiple ES (for further examples, see also McRoberts *et al.* 2012). In the research context, Cardinale *et al.* (2012) outlined two most important directions to be undertaken: (i) detailing the mechanistic links between ecosystem functions and services; and (ii) developing theoretical approaches that can link the small-scale research, (mechanistic focus of biodiversity and ecosystem functioning) to large-scale patterns that are the focus of biodiversity and ES.

FES research in Italy

As obtained in RS D, the contribution to ES and FES research from Italy is very scarce (25 and 8 publications from 2000 to 2012, respectively; see Table 8). The earliest papers focusing on ES from Italy were released in 2003 (Horton *et al.* 2003), about 10 years later than those already available at global scale in early 90's (e.g. Costanza and Daly 1992). Considering FES, the delay is similar. Moreover, FES-related papers are mainly focused on the "ES Assessment" and the "Economics of ES" TAs (see Figure 11). Indeed, the most-cited references for RS D is Horton *et al.* (2003), which is mainly focused on the willingness to pay for environmental services, and not specifically with regards to ecological and societal aspects of forest ecosystems (see Table 10). Taking into account the results from RS E, research projects that totally address the FES topic are still scarcely available at continental scale (see Appendix 1). Results appear incomplete. Indeed, some important European research pathways about the ES topic were not founded through RS E. For example, this is the case of the RUBICODE ("Rationalizing Biodiversity Conservation in Dynamic Ecosystems", on line at: www.rubicode.net) project, which ended on 2009, and collated and reviewed information on ES for the main terrestrial and freshwater ecosystems in Europe in order to provide a framework to rationalize biodiversity conservation strategies (Harrison 2010; Anton *et al.* 2010). Therefore, since the "EU Biodiversity Strategy" (EU-BS) (European Commission 2011), the importance of mapping and assessing ES have gained more attention among scientists, up to the establishment of the MAES (Mapping and Assessment of Ecosystems and their Services)

Working Group with the main objective to support Member States in fulfilling the requirements of Action 5[4] of the EU-BS (Maes *et al.* 2013b). Italy was not involved in the pilot studies phase (to be completed for the end of 2014). Of course, RS E does not consider FES-related projects currently at work at national level in Italy. However, a deeper analysis indicates that several projects on FES are currently at work at national level in Italy. Some examples are: (i) the INTEGRAL ("Integrated management of European Forest Landscapes"; on line at: www.integral-project.eu) project, which is specifically oriented to diminish the discrepancies between policy and management approaches in improving the potential of European forest landscapes to deliver multiple services, as well as to provide management guide-lines according to the ecological and socio-economic contexts; (ii) the MIMOSE ("Development of innovative models for multiscale monitoring of ES indicators in Mediterranean forests") project, which is conceived to build and implement a set of spatially-explicit indicators for mapping and valuing ES for the Mediterranean forests (Chirici *et al.* 2014); and (iii) the LIFE+ MGN ("Making Good Natura"; on line at: www.lifemgn-serviziecosistemici.eu) which is aimed at developing innovative approaches of environmental governance to preserve agro-forest-ecosystems, as well as elaborating instruments for qualitative and quantitative valuation of the ES in the study sites of the Natura 2000 network.

Considering the above-mentioned issues and the recent huge efforts by EU (e.g. Kumar 2010) and its Member States to implement the ES approach into development strategies, Italy does not have its own proposal yet (Brouwer *et al.* 2013). So far, applied research on forests and other ecosystems (including the services they provide) has suffered from the scarcity of data availability, the fragmentation and differentiation of both on-ground and remote-sensed information, the weakness of a trans-disciplinary cooperation between Universities, National Research Institutes, and local Administrative Bodies at national level, and the sensible reduction of economic investments in research, innovation and development (-1.6% between 2011 and 2012; further details are available on line at: http://www.istat.it/it/archivio /105810).

Challenges for forest ecosystem services research

At conclusion, the review, both focused on literature and projects, gave an opportunity to deeper understand the current lacks of information, issues, and future challenges about the FES-related research from European to Italian level. Since the biodiversity is considered a baseline for the ecosystem functioning (i.e. health and vitality) and resilience, research horizons have to be targeted on reducing the gap between the assessment of biodiversity conservation state (mostly ecology-based) and the evaluation of the other ES (mostly economy-based). More specifically, the fundamental role of forest ecosystems in ameliorating the human well-being needs to be deeper investigated and understood at local level (e.g. the effects forest bathing on human health), especially with regards to the linkage between the ecosystem

[4] "Action 5 of the EU Biodiversity Strategy requires Member States, with the assistance of the Commission, to map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020".

processes, the services provided, and the whole environment correlated (i.e. changes in and between land use classes, Nagendra *et al.* 2004). For example, the improvement of knowledge about forest ecosystems and their services can be realized through concentrating research efforts on modeling and mapping ES fluxes (from sources to beneficiaries), from which the economic values strictly depends (Abson and Termansen 2011).

Although this challenge generally regards the whole scientific community at global scale, it is particularly amplified for the Italian context. Indeed, our results demonstrated that, in the context of ES in general, and of FES in particular, there is a huge gap (in terms of number of publications, and amount of participations in project consortia) between Italian contributions (in terms of research impact, outcomes and results) and those available at global level. This appears not completely consistent with the important role of forest resources for providing ES in Italy, ranging from biodiversity conservation, to the preservation of cultural and spiritual heritages, and to the contribution to economic incomes in many rural and marginal communities (see MIPAAF *et al.* 2008). Therefore, other challenges for FES research stand in improving the interchange of knowledge between researchers, scientists, experts, and technicians, and local communities, as well as in an effective involvement of stakeholders' needs into decision-making processes (see also Fisher *et al.* 2008). A complete understanding of forests and their services is not only a consequence of analysis and simulations of the related processes, but also a continuous assessment of the needs of people living closely to natural resources (see also Bormann *et al.* 2007).

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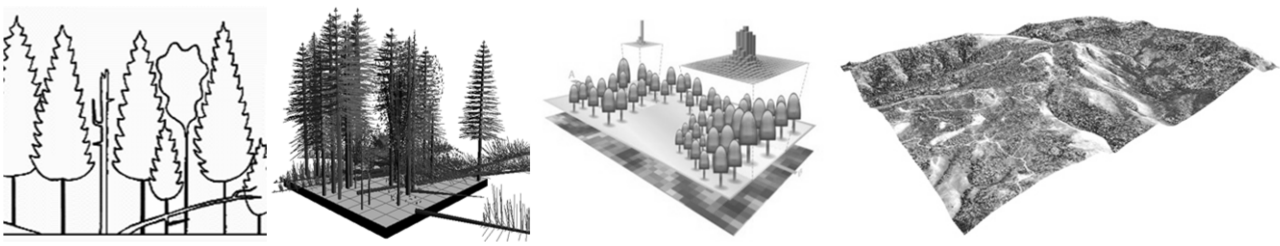
3

Modeling forests for multiple services: advanced approaches and recent techniques



www.na.fs.fed.us; www.fs.usda.gov/ccrc/tools/fvs; Seidl *et al.* (2013); spie.org

Forest modeling is particularly useful to simulate forest landscape dynamics, and long term consequences of climate change impacts or management practices. To date, several modeling techniques have been used to support multi-purpose forestry from local to global scale. More recently, forest ecosystem models have become the core of decision support systems for sustainable forest management, thus leading forest managers to solve even more complex decisions and to deeper understand future-oriented natural dynamics and increasing environmental changes.

This chapter provides an overview of some currently available forest ecosystem models and decision support systems for forest management, with a particular focus on how to implement them to assess the impact of forest management on services provision (including the use of indicators, and mapping techniques), as well as on how to use them for supporting decision-making processes at different planning scales. Two case studies are presented, accordingly. The first one is focused on the implementation of a semi-automatic algorithm to map forest ecosystem functions in a Natura2000 Network area in Central Italy. The second one is specifically oriented to simulate forest ecosystem services provision in three Italian landscapes, according to alternative future-oriented scenarios.

3.1 Tools and approaches for modeling and mapping forest ecosystem services

3.1.1 The role of decision-support systems in forest management

With its emphasis on broad, holistic, integrated perspectives, the concept of forest ecosystem management posed serious new challenges to the delivery of effective decision support (Rauscher 1999; Schmoldt and Rauscher 1996). As a consequence, numerous expert systems were developed to assist with forest pest management, silvicultural prescriptions, and timber harvesting, among other things (Durkin 1993). Indeed, simulation and optimization algorithms have been included in software to guide forest managers since the 1960s, and now at least 100 computerized decision support systems (DSS), with various levels of sophistication, have been developed and are being widely used in numerous countries (Eriksson and Borges 2014). A DSS is "a computer-based system composed of a language system, presentation system, knowledge system, and problem-processing system whose collective purpose is the support of decision-making activities" (Holsapple 2003, p. 551). DSS generally implement the Analytic Hierarchy Process (AHP) and similar Multi-Criteria Decision Analysis (MCDA) methods, knowledge-based systems that provide a framework for applying procedural or reasoning knowledge to decision problems and, perhaps some more arguably, optimization systems (Reynolds 2005; Kangas *et al.* 2008). Figure 12 reports the general architecture of a DSS.

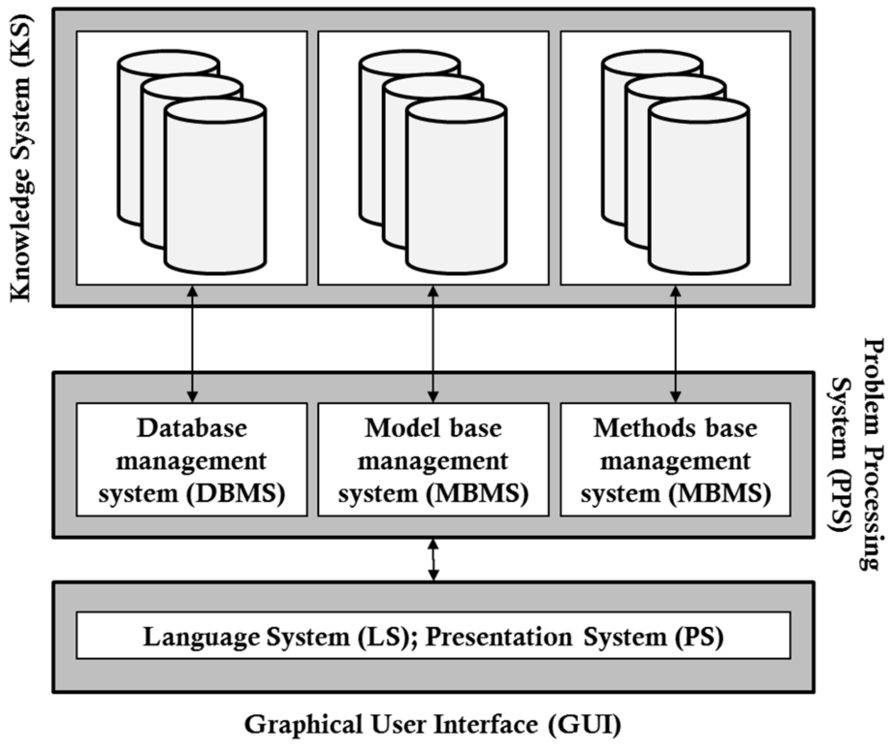


Figure 12: DSS architecture (Eriksson and Borges 2014, modified).

Several reviews of forest DSS were presented in the 2000s. Johnson *et al.* (2007) characterized 32 systems according to the decision-making factors they considered (e.g.

biodiversity indicators supported, forest disturbances, silviculture, etc.) and included 15 in-depth studies of successes and failures of DSS applications. Johnson *et al.* (2007) also cited reviews of DSS capabilities to assist with National Forest Plans (Schuster *et al.* 1993), ecosystem management (Mowrer 1997, Rauscher 1999), and biodiversity in county-level planning (Johnson and Lachman 2001). Reynolds *et al.* (2008) reviewed 10 systems. More recently, Borges *et al.* (2014) reported a large-scale survey on the current availability of DSS world-wide, as final outcomes of the COST Action "Forest Management Decision Support Systems" (FORSYS). Table 11 reports the number of forest DSS for each problem type as available world-wide.

Table 11: Number of DSS by specific characteristics and problem type cluster (Borges *et al.* 2014, modified).

	Stand, long term, supplying only wood products	Stand, long term, not supplying only wood products	Stand, medium or short term, supplying only wood product	Stand, medium or short term, not supplying only wood products	Forest, long term supplying only wood products	Forest, long term not supplying only wood products	Forest, medium or short term, supplying only wood products	Forest, medium or short term, not supplying only wood products	All regional types
DSS (Total)	23	15	20	15	29	58	26	31	48
With database	14	9	13	12	20	53	24	28	43
With GIS	4	6	3	12	12	49	19	24	39
With other KM	3	8	3	5	5	32	13	12	30
With Vegetation Simulator	18	15	16	7	23	54	17	9	26
With automated solution quantitative support	7	10	12	10	21	53	26	26	30
With automated solution qualitative support	2	4	2	7		5	3	7	5
With explanatory support for participatory process				2		10	3	7	13
DSS Users									
Research	15	9	9	7	21	39	8	11	23
Consultant	7	5	6	1	23	27	13	2	7
Managers	11	6	9	5	12	34	19	15	13
Public				3		4	3	5	7
Other (e.g. students)	8	4	6	2	6	12	4	1	8

Borges *et al.* (2014) pointed out that: (i) in general, the use of computerized tools to support forest management planning is pervasive; (ii) the use of DSS is widespread and mainly oriented to address long-term planning issues and timber demands; (iii) although most of planning problems are perceived as spatial problems, a low percentage of DSS includes GIS; (iv) although Menzel *et al.* (2012) described the potentialities of DSS in participatory planning processes, the information regarding the DSS development is very scarce (e.g. no active participation of local communities in the development of a DSS).

3.1.2 Mapping changes of forest ecosystems and their services

Land use (LU) activities have transformed a large portion of the planet's land surface (Foley *et al.* 2005). Several decades of research have revealed the environmental LU impacts throughout the globe, ranging from changes in atmospheric composition to the extensive modification of Earth's ecosystems (Vitousek *et al.* 1997; Matson *et al.* 1997; Tilman *et al.* 2001; Wackernagel *et al.* 2002). More recently, Ellis *et al.* (2010) highlighted that, by 2000, most of the terrestrial biosphere was transformed into predominantly anthropogenic ecological patterns combining lands used for agriculture and urban settlements and their legacy; the remnant, recovering and other managed novel ecosystems embedded within Anthromes (see also: Ramankutty and Foley 1999; Ellis and Ramankutty 2008). Thus, land use cover and change (LUCC) directly (Lambin *et al.* 2001): (i) impact biodiversity worldwide (Sala *et al.* 2000); (ii) contribute to local and regional climate change (Chase *et al.* 2000) as well as to global climate warming (Houghton *et al.* 1999); (iii) are the primary source of soil degradation (Tolba *et al.* 1992); and (iv) by altering ES, affect the ability of biological systems to support human needs (Vitousek *et al.* 1997). Evidence of the ES reduction owing to LUCC is gradually accumulating (Martínez *et al.* 2009), especially in the case of pollination services (Priess *et al.* 2007; Ricketts *et al.* 2008; Steffan-Dewenter and Westphal 2008), carbon storage (Huston and Marland 2003; Kirby and Potvin 2007); hydrology (Strange *et al.* 1999), and climate change (Schröter *et al.* 2005), among the others. For a more detailed description of the human-induced effects on forest ecosystem resilience and related services, the reader is referred to Chapter 2.

In order to plan actions to slow rates of decline, secure future ES provision for human use and forest investment in ecosystem management, a unified and global ecosystems risk assessment framework was published by Keith *et al.* (2013), who proposed the IUCN Red List of Ecosystems, according to the approach followed for the preparation of the IUCN Red List of Threatened Species (Rodrigues *et al.* 2006). Considering the effects of LUCC on forest extent, Rudel *et al.* (2005) noted that a combination of changing bio-physical and socio-economic conditions in the Mediterranean basin over a period of centuries contributed to gradual declines in forest cover with no recovery until the last three decades of the 20th century. In particular, Marchetti *et al.* (2012) described in Italy an increment of forest cover of about 512,000 hectares from 1990 to 2008.

Two theories underline the forest transitions dynamic. On the other hand, forest transitions occur because farmers discover over the time their most productive lands, concentrate production on them, and abandon their least productive lands which then revert to forest (Mather 2007). In addition, forest transitions dynamic involves the concept of 'leakage', such as a displacement of deforestation to neighboring locations through migration of agents of deforestation or through trade in timber or agricultural products (Meyfroidt *et al.* 2010).

Despite the considerations about trends and effects of LUCC over the time, policy, management and land planning urgently require spatial analysis of ES at global (Naidoo *et al.* 2008), continental (Metzger *et al.* 2006; Kienast *et al.* 2009) and regional (Chan *et al.* 2006; Egoh *et al.* 2009; Eigenbrod *et al.* 2010) scale (Carpenter *et al.* 2009). Therefore, there is an increasing need for mapping the simultaneous provision of multiple ES at landscape scale (see Naidoo and Ricketts 2006), and for modeling LUCC (Verburg *et al.* 2009). ES maps created by modeling the relationship between samples of a service and readily measurable environmental variables (i.e. climate, land cover, soil types) are more common within the ES literature (Eigenbrod *et al.* 2010). They were used in large-scale multi-service studies to map carbon storage (e.g. Milne and Brown 1997; Eigenbrod *et al.* 2009), carbon fluxes (McGuire *et al.* 2001), and biodiversity priority areas (Chan *et al.* 2006). Proxy-based maps are more common than maps based on primary data (e.g. Sutton and Costanza 2002; Chan *et al.* 2006; Troy and Wilson 2006; Turner *et al.* 2007; Egoh *et al.* 2008). At this point, it is important to note that when mapping multiple ES at different scales and by many spatial attributes, forests (and other natural systems) are generally considered as integrated parts in a more broadly holistic approach, which considers ecosystems as reflections of dynamic change, disturbance, and non-equilibrium conditions (e.g. Pickett *et al.* 1992, among the others). Consequently, Raffaelli and Frid (2010) pointed out that a holistic perspective toward ES provides insights into the many unexpected consequences of human activity. A detailed and more comprehensive explanation about ES interactions, assessment scales, trade-offs and environmental management can be found in Menzie *et al.* (2012).

As also summarized by Nelson and Daily (2010), Table 12 reports the main characteristics of the tools currently available at global level for mapping and assessing a complete set of ES (and FES) by an integrated perspective.

Table 12: Summary table concerning the currently available ES mapping tools, including a brief description and the most important references.

Acronym	Complete name	Brief description	Main reference	Web link
InVEST	Integrated Valuation of Ecosystem Services and Tradeoffs	InVEST determines ES provision and value at a point on the landscape by using ecological and economic production functions, where LULC and related management and biophysical data at the point and elsewhere on the landscape are inputs.	Nelson *et al.* (2009)	www.naturalcapitalproject.org/InVEST.html
ARIES	The Artificial Intelligence for Ecosystem Services	ARIES uses a benefit-transfer approach. Under this methodology, each point on the landscape is assigned ES provision and value largely according to its LULC, where the ecosystem service provision and values associated with the LULC are culled from other site-based studies.	Villa *et al.* (2014)	www.ariesonline.org/
MIMES	The Multi-scale Integrated Models of Ecosystem Services	MIMES is a suite of models for land use change and marine spatial planning decision making. The models quantify the effects of land and sea use change on ecosystem services and can be run at global, regional, and local levels.	Boumans and Costanza (2007)	www.ebmtools.org/mimes.htmll
ENVISION	N/A	ENVISION is a GIS-based tool for scenario-based community and regional planning and environmental assessments. ENVISION combines a spatially-explicit polygon-based representation of a landscape, a set of application-define policies (decision rules) that are grouped into alternative scenarios, landscape change models, and models of ecological, social and economic services to simulate land use change and provide decision-makers, planners, and the public with information about resulting effects on indices of valued products of the landscape.	Hulse *et al.* (2004)	envision.bioe.orst.edu/

3.2 The indicator-side of forest ecosystem services

3.2.1 How to measure ecosystem services: towards the use of indicators

In general, measuring ES has to face with several constraints (Patterson 2011), such as: (i) how to separate the concept of 'stock' from 'flow' (the interest that is generated from account over a given period of time); (ii) how to link a particular action or intervention on landscape with a predetermined consequent reaction in ES, and how to account for the variety of beneficiaries that will be affected, or the length of time that impact will endure; and (iii) how to distinguish the ecosystems production that can be potentially used and the production that is currently used or collected from people. Nevertheless, ES can be assessed at different stages of production by measuring the generation of ecosystem processes, by quantifying the magnitude of attributes or intermediate service levels, or by assessing the amount of final service benefit (Brauman *et al.* 2007).

Among the various existing methods to assess ES, indicators are measurable surrogates for environmental end points (such as biodiversity) that are assumed to be of value to the public (Noss 1990). Ideally, an indicator should be (Noss 1990): (i) sufficiently sensitive to provide an early warning of change; (ii) distributed over a broad geographical area, or otherwise widely applicable; (iii) capable of providing a continuous assessment over a wide range of stress; (iv) relatively independent of sample size; (v) easy and cost-effective to measure, assess, assay, and/or calculate; (vi) able to differentiate between natural cycles or trends and those induced by anthropogenic stress; and (vii) relevant to ecologically significant phenomena (Cook 1976; Sheehan 1984; Munn 1988). An indicator is "a measure, generally quantitative, that can be used to illustrate and communicate complex phenomena simply, including trends and progresses over time" (EEA 2005). An indicator provides a clue to a matter of larger significance or makes perceptible a trend or phenomenon that is not immediately detectable. An indicator is a sign or symptom that makes something known with a reasonable degree of certainty. An indicator "reveals, gives evidence, and its significance extends beyond what is actually measured to a larger phenomenon of interest" (IITF 2000). In assessing ES, comprehensive sets of indicators are needed, and they have to be selected according to the ecosystem properties, functions, and services (see e.g. van Oudenhoven *et al.* 2012; Syrbe and Walz 2012; Burkhard *et al.* 2012a). Moreover, the indicators should be clear and understandable, enabling communication between scientists and stakeholders (Burkhard *et al.* 2012b). The selection of indicators should be based on robust procedures and guidelines (e.g. van Oudenhoven *et al.* 2012; Koschke *et al.* 2012; Haines-Young *et al.* 2012).

Studies concerning ES indicators are manifold around the world. Recently, Hernández-Morcillo *et al.* (2013) scientifically recognized frameworks to develop a holistic understanding of how cultural services indicators are conceived within ES research. Shoyama *et al.* (2013) evaluated public preferences for biodiversity conservation and climate-change mitigation policies in Japan, adopting explicit indicators. Ausseil *et al.* (2013) developed spatially-explicit models of indicators of important ES in New Zealand, thus assessing the change of such indicators with regards to two particular extremes. Comprehensively, Müller and Burkhard

(2012), and Kandziora *et al.* (2012) investigated the main interrelations between the ES concept and the Driver-Pressure-State-Impact-Response (DPSIR) approach. At European scales, Haynes-Young *et al.* (2012) developed an approach for mapping indicators of the ecosystem potentiality to supply ES, and the impact of LUCC upon them. At global level, Layke *et al.* (2012) presented an evaluation of ES indicators, which was compiled from over 20 ecosystem assessments conducted at multiple scales and in many Countries. Burkhard *et al.* (2011) carried out an interesting work on building hypotheses on the development of (temperate forest) ecosystem features during the different phases of the adaptive cycle, thus proponing several potential indicators about each ecosystem orientor (i.e. thermodynamics, information, networks, eco-physiology, dynamics, and ecosystem services). More generally, Niemeijer and de Groot (2008) proposed a selection of indicators, which was based on the enhanced DPSIR framework.

3.2.2 Building a unified framework of forest ecosystem services indicators

In this section, the FES indicators framework (FES-IF) is formulated for the following purposes: (i) to easily combine FES with the ecological processes involved and the available approaches for their assessment; (ii) to provide an overview of models and methods applied for measuring FES; and (iii) to establish a common basis for a better understanding on how goods and services are delivered by forest resources. Methodologically, FES-IF follows a process-based approach. In order to be as exhaustive as possible, FES-IF is based on currently available literature concerning the FES assessment through the use of indicators (e.g. Dale and Beyeler 2001; de Groot *et al.* 2002; Jørgensen and Xu 2010; de Groot *et al.* 2010; Burkhard *et al.* 2011; Haines-Young and Potschin 2010; Kandziora *et al.* 2012). At first, FES Classes and Types have been identified and structured. Then, the most important indicators have been identified, explained and linked to each FES. Secondly, the main forest ecosystem processes that are directly involved in the provision of FES have been found. Finally, for each FES indicator, the methods applied for its calculation, and the minimum assessment scale (MAS) are provided. Table 13 reports the proposed FES-IF. Within the FES-IF, no specification about the use and implementation of indicators is given, neither for their suitability nor reliability.

Table 13: Summary table concerning the FES-IF. The table reports: (i) the FES classes and types, as hereby classified; (ii) the main ecosystem process involved in the generation of the service; (iii) the FES indicators; (iv) the minimum assessment scale (MAS) as the minimum geographic level at which the FES can be quantified and evaluated using the related FES indicator; (v) an indication of the models, methods and approaches can be applied for assessing a given FES, or for implementing the selected indicator; (vi) the indicative references, as examples of the studies using and implementing the outlined methods and approaches for the FES assessment.

[1]Models, methods and approaches to be applied for assessing three different FES (such as climate change mitigation, the control of hydrological processes and the air quality regulation) can be easily confused and interchanged, because they concern many common ecosystem processes, most of them involving climate, atmosphere, water, soil and vegetation interactions. [2]The indicative references are the most important sources of information as identified and chosen to describe and explain the models, methods and approaches that can be used for assessing FES.

FES Class (es)	FES Type (s)	Forest ecosystem processes involved	FES Indicator (s)	Minimum Assessment Scale (MAS)	Models, methods and approaches applied	Indicative references[2]
Provisioning services	Timber production	Natural growth, competition and seed dispersal, presence of tree species with potential use for timber, fuel or raw materials	Total amount of timber harvested or total amount of biomass to be used for timber production (and market allocation)	Local (Forest stand) level	Almost all currently available forest ecosystem models can predict and simulate forest growth, planned forestry interventions, timber productivity and harvesting	For exhaustive reviews on forest ecosystem models, see: Bugmann (2001); Pacala *et al.* (1996); Portè and Bartelink (2002); Baker (1989)
					Forest-wood-energy chain simulator	Ziesak *et al.* (2004)
	NWFPs (fruits, berries, truffles, mushrooms, etc.)	Natural conditions and potentialities (plant species availability, soil and climate) for natural production of forest goods and products different from timber	Total amount of edible forest products, in terms of quantity picked or consumed (for each NWFPs category)	Local (Forest stand) level)	Prediction and modeling of the production of non-wood forest goods (and other services)	For a complete review concerning Mediterranean forests, see Palahi *et al.* (2009)
	Fresh water supply	Presence of natural water reservoirs, role of forests in water infiltration and its gradual and healthy releases	Total amount of water bodies (and related areas) within a forest landscape; total amount of	Landscape (forested watershed) level	Water-Global Assessment and Prognosis (WaterGAP)	Döll *et al.* (1999); Alcamo *et al.* (2000)
					WaterGAP Global Hydrology Model	Döll *et al.* (2003)

FES Class (es)	FES Type (s)	Forest ecosystem processes involved	FES Indicator (s)	Minimum Assessment Scale (MAS)	Models, methods and approaches applied	Indicative references[2]
			freshwater consumed by people and/or by communities living close to forest		(WGHM)	
					Soil and Water Assessment Tool (SWAT)	Schuol *et al.* (2008); Faramarzi *et al.* (2009)
Regulating services	Climate change mitigation[1]	Potentialities of tree species and forest cover to influence and mitigate climate change though intrinsic eco-physiological processes and by the soil-plant-atmosphere interchanges	Greenhouse gas balance (e.g. carbon stocked and released)	Individual tree level	FORUG	Verbeeck *et al.* (2006)
					FORECAST	Seely *et al.* (2002)
					BIOME-BGC	Running (1993); Schimel *et al.* (2000); White *et al.* (2000)
					CO_2FIX model	Mohren and Klein-Goldewijk (1990); Masera *et al.* (2003); Schelhaas *et al.* (2004); Nabuurs and Schelhaas (2002)
					YASSO forest soil model	Liski *et al.* (2005)
					FullCAM	Paul *et al.* (2013)
					CBM-CFS3	Kurz *et al.* (2008); Kurz *et al.* (2009)
	Control of hydrological processes[1]	Role of forest vegetation in soil formation, movement, and retention (including prevention for floods, avalanches, and other runoff-generated events)	Erosion rate; surface geology; land cover; sediment yield; root-matrix and depth	Landscape (forested watershed) level	Pacific Southwest Inter-Agency Committee (PSIAC) and Modified Pacific Southwest Inter-Agency Committee (MPSIAC) family models	Daneshvar and Bagherzadeh (2012); Safamanesh *et al.* (2006)
					Soil and Water Assessment Tool (SWAT)	Amatya and Jha (2011)
					DRAINMOD	Skaggs *et al.* (2012); Tian *et al.* (2012)
					Riparian Ecosystem	Lowrance *et al.* (2000);

FES Class (es)	FES Type (s)	Forest ecosystem processes involved	FES Indicator (s)	Minimum Assessment Scale (MAS)	Models, methods and approaches applied	Indicative references[2]
					Management Model (REMM)	Inamdar *et al.* (1999); Liu *et al.* (2007)
					Universal Soil Loss Equation (USLE), Modified Universal Soil Loss Equation (MUSLE), and Revised Universal Soil Loss Equation (RUSLE)	Wischmeier and Smith (1978); Williams (1975); Renard *et al.* (1991)
					Water Erosion Prediction Program (WEPP)	Dun *et al.* (2009)
					TOPOG	O'loughlin (1986)
	Regulation of natural hazards	Resilience and stability characteristics of forest ecosystems to control and reduce the impacts of wind, wildfires, pests, and other natural disasters (not directly derived from hydrological processes) for human population		Local (Forest stand) level	Mechanistic, analytical or empirical models	For a complete review, see: Hanewinkel *et al.* (2011)
	Air quality regulation[1]	Capacity of forests to extract chemicals and aerosols from the atmosphere	Leaf area index; amount of air pollutants removed or fixed	Individual tree level	i-Tree Eco model	www.itreetools.org; Hirabayashi *et al.* (2012)
					Multi-layered model (MLM) for O3 uptake	Launiainen *et al.* (2013)
					CHIMERE (air quality model)	Bessagnet *et al.* (2004); Alonso *et al.* (2011)
					Community Multi-scale Air Quality (CMAQ) model	Byun and Schere (2006)
					French national air quality forecasting and monitoring system (PREV'AIR)	Honoré *et al.* (2008)

FES Class (es)	FES Type (s)	Forest ecosystem processes involved	FES Indicator (s)	Minimum Assessment Scale (MAS)	Models, methods and approaches applied	Indicative references[2]
					Urban forest effects model (UFORE)	Currie and Bass (2008)
Supporting services	Conservation of biological diversity	Natural asset of forest biota, species richness, composition, variety of wildlife	Tree species richness; forest naturalness; red list of threatened species; number of tree alien species (inversely)	Local (Forest stand) level	Models to assess biodiversity itself (FOR-biodiversity) and/or Models using components of biodiversity to assess 'environmental health' (FROM-biodiversity) (Duelli and Obrist 2003)	Lindenmayer *et al.* (2000); Larsson (2001); Feld *et al.* (2009); Failing and Gregory (2003); Noss (1990); Ojea *et al.* (2010)
	Preserving habitat integrity and landscape fragmentation	Intrinsic ecological integrity, natural colonization and re-sprouting, environmental adaptability, plant communities stability, refuges for wildlife, potentialities for naturalness	Habitats cover; mean patch sizes; forest landscape patterns	Landscape level	Models to evaluate forest landscape patterns, habitat integrity and connectivity, as well as forest naturalness	Schumaker (1996); Brooks *et al.* (1998); Carignan and Villard (2002); McRoberts *et al.* (2012);
	Gene-pool and gene-flows protections	Presence of tree species with (potentially) useful genetic materials	Number of tree species protected for genepool and preserved for seed production	Landscape level	Models assessing the number of forest alien species in endangered forests	Wittenberg and Cock (2001); McGeoch *et al.* (2010)
Cultural services	Aesthetic appreciation, historical identity, recreational opportunities	Presence of forest ecosystems/landscapes characteristics with a particular importance for tourism, recreational and educational activities; forests representing a historical heritage; aesthetic value of the forest landscape	Number of visits; number of sites with particular cultural features; scenic beauty of forest landscapes (e.g. scoring)	Landscape level	Analytic Hierarchy Process (AHP) for natural attractions evaluation method	Deng *et al.* (2002); Kajanus *et al.* (2004);
					Tourism Features Simulator (TFS)	Walker *et al.* (1998)
					Scenic Beauty Estimation (SBE) method	Arthur (1977); Hull and Buhyoff (1986); Daniel (2001)
					Actual Tourist	Scrinzi and Floris (2000)

FES Class (es)	FES Type (s)	Forest ecosystem processes involved	FES Indicator (s)	Minimum Assessment Scale (MAS)	Models, methods and approaches applied	Indicative references[2]
					Recreational Attendance (ATT) model	
					Approaches for assessing sustainability of tourism-related activities	Lee (2007) Choi and Sirakaya (2006)

3.3 Case study 2: The implementation of a multilevel algorithm to map forest ecosystem functions in a Natura2000 site in Italy[5]

3.3.1 The context

In last decades, numerous efforts have been made at global level to implement the concept of sustainability into forest resources management planning, and many initiatives took place to define the Sustainable Forest Management (SFM) and to develop tools supporting it (Hickey *et al.* 2005). In this sense, different international progresses have been made in order to adopt and implement the SFM concept into policy making processes, from continental to forest unit level (Wijewardana 2008). In Europe, the first initiative adopting SFM was led by the Ministerial Conference on the Protection of Forests in Europe (MCPFE, now Forest Europe) in Helsinki in 1993 (MCPFE 1993). Since then, different guidelines to correctly implement SFM in forest management planning were developed (MCPFE 2003), and as a consequence different levels of forest management planning were implemented at national level as well (Cullotta and Maetzke 2008). Furthermore, numerous efforts have been done in recent years by scientific community in order to support the forest decision making processes stressing the multi-functional role of forests (Wolfslehner and Vacik 2011; Lexer and Brooks 2005; Stenger *et al.* 2009; Wang *et al.* 2010; Rametsteiner *et al.* 2011; Gatto *et al.* 2009; Daily and Matson 2008). Indeed, forests: (i) were and are exploited for both timber and non-timber products everywhere; (ii) give protection against very different types of hazard, torrents and avalanches in the mountains, soil erosion by water and wind, contamination of ground and spring water, desertification, etc.; (iii) are increasingly used by urban populations for recreational purposes; (iv) represent the habitat of a considerable part of our flora and fauna, which must be sustained for the conservation of biodiversity (Führer 2000).

Thus, sustainable forest management and planning are fundamental tools to ensure forest ecosystem health and productivity, and as a consequence the continue provision of goods and services to local communities (Bray *et al.* 2003). Theoretically, the forest multi-functionality has a broader task. In fact, when forest management planning must take into account new ranges of spatial and temporal scales in which natural or human phenomena take place (Sverdrup and Stjernquist 2002), and where other forest functions such as ecological or social functions develop with spatial continuum, the forests have to be seen and integrated into more generic concepts such as the landscape or the watershed, in which they interact as partners (Farcy 2004). For this reason, the forest management planning at landscape level represents a sound approach that takes into account all forest functions in an integrated and holistic way (Kangas and Store 2002).

In Italy, the Forest Landscape Management Planning (FLMP) places itself at an intermediate level in the forest management planning hierarchical framework defining peculiar functions of forests to be planned (Cullotta and Maetzke 2008), thus representing an integrated tool particularly useful to address the long-term forest management issues, with a specific

[5] Source: Vizzarri *et al.* (2014a).

attention to those forest features that cannot be systematically considered when working at the stand level (Cantiani *et al.* 2010). FLMP provides different forest management guidelines according to SFM principles (Secco *et al.* 2006) and distributes them in the space and time. So far, FLMP (or a similar planning framework) has been successfully adopted in several pilot studies in Italy (Cullotta and Maetzke 2008), such as: (i) the Cadore, Longaronese, and Zoldo MC (Portoghesi *et al.* 2012); (ii) the Asiago plateau MC (Corona *et al.* 2010); (iii) the Piné plateau (ongoing process); (iv) the watershed of Trasimeno lake; (v) the Trigno-Biferno rivers MC; (vi) the Alto Molise MC; (vii) the Agri river plain area and the "Appennino Lucano-Val d'Agri-Lagonegrese" National Park (ongoing process); (viii) the Collina Materana MC (Cantiani *et al.* 2010; Paletto *et al.* 2012); (ix) the Natural Reserve of Sosio valley and Palazzo Adriano mountains; (x) the North-western area of Etna mountain; and (xi) the "Arci-Grighine" district (Paletto *et al.* 2011). Moreover, the FLMP implementation in the regional forest planning framework can be found e.g. in Lombardia and Piemonte regions. Figure 13(a) shows the location of the above-mentioned FLMP pilot studies and regional implementations.

3.3.2 Objectives and methodology

This study proposes a methodological approach to identify and map specific Functional Destination Units (FDUs) through a FLMP approach. FDUs are intended here as forest areas providing the same forest ecosystem function (Führer 2000). Particularly, the aim is threefold: (i) assessing forest ecosystem services in the context of forest management planning at landscape level; (ii) understanding how remotely-sensed and inventory data can be jointly implemented to map forest ecosystem functions at landscape level; and (iii) mapping forest ecosystem functions to improve the usefulness of forest management planning at landscape level.

The study area belongs to the Natura2000 Network Site of Community Importance (SCI) "La Gallinola, M.teMiletto, M.ti del Matese" (IT7222287). It is located in Molise region, Central Italy, and covers an area of 25,160 ha (see Figure 13(b)), of which about 17,300 ha are forested. The natural landscape of Matese SCI is extremely diversified and patched (Garfì and Marchetti 2011). Pastoral areas alternate with forests and farmlands. According to the European Forest Types (FTs) framework (EEA 2006; Barbati *et al.* 2007), the most representative FTs are: (i) the European beech forest (about 8,000 ha); (ii) the hop-hornbeam forest (about 2,300 ha); and (iii) the Turkey oak forest (2,162 ha) (Garfì and Marchetti 2011).

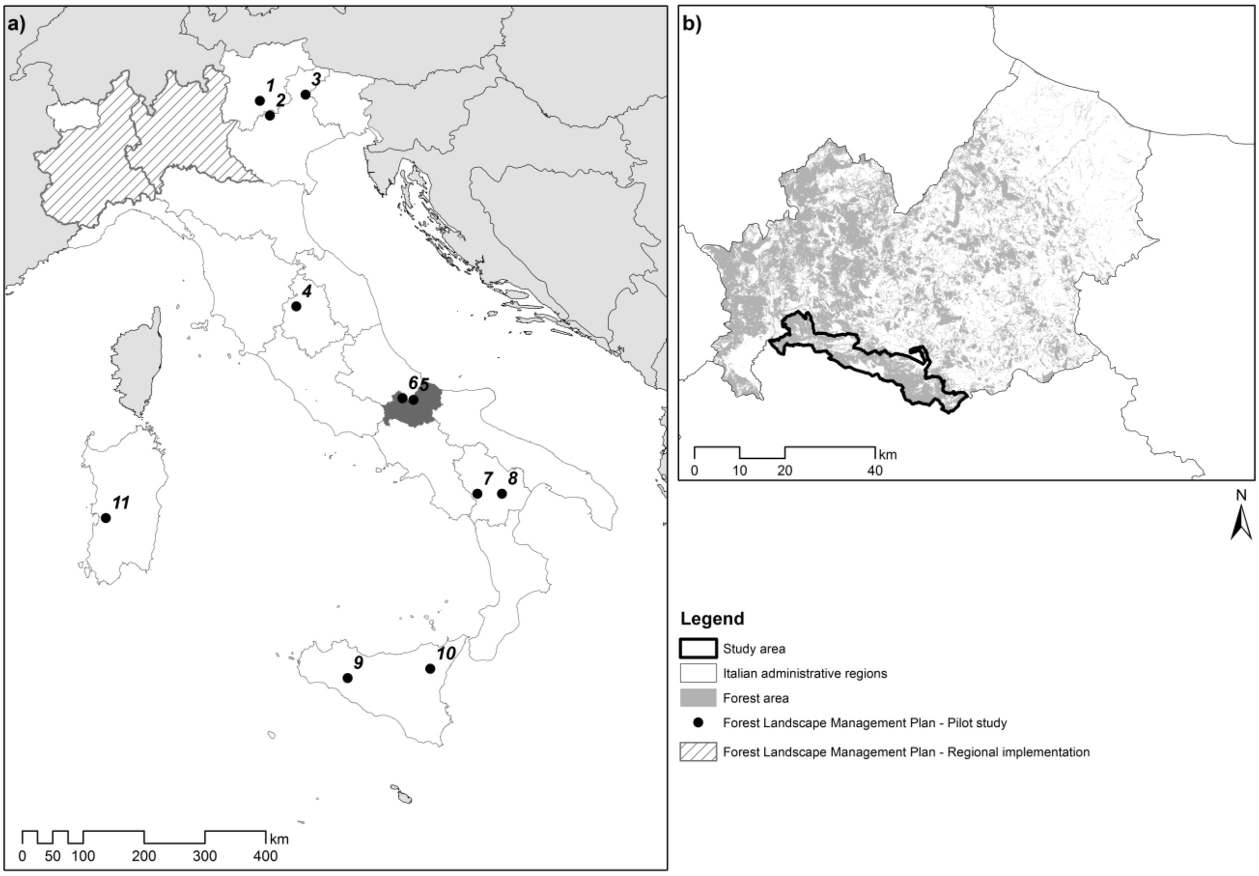


Figure 13: FLMP pilot studies and regional implementation distributions in Italy (a), and a zoom-on both Molise region and case study area (b). Numbering of FLMP pilot studies as follows: (1) the Cadore, Longaronese, and Zoldo MC; (2) the Asiago plateau MC; (3) the Piné plateau; (4) the watershed of Trasimeno lake; (5) the Trigno-Biferno rivers MC; (6) the Alto Molise MC; (7) the Agri river plain area and the "Appennino Lucano-Val d'Agri-Lagonegrese" National Park; (8) the Collina Materana MC; (9) the Natural Reserve of Sosio valley and Palazzo Adriano mountains; (10) the North-western area of Etna mountain; and (11) the "Arci-Grighine" district. References in the text.

The process for mapping FDUs is structured into the following steps: (i) the definition of the forest ecosystems functions to be assigned during the inventory phase; (ii) the collection of the forest stand parameters during the inventory phase; (iii) the selection of the main forest attributes correlated to the selected forest ecosystem functions and the use of a k-NN method for their estimations; (iv) the implementation of a MCML approach to map the selected FDUs; (v) the agreement assessment of the final FDUs map.

Definition of forest ecosystem functions

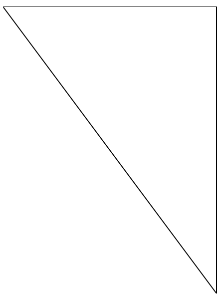
Before the field survey phase a list of main forest functions was identified by a panel of local forest technicians and plant experts with a high knowledge of the study area. In this phase, functions were initially defined as: 'the capacities of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly' (de Groot and Wagenaar-Hummelinck 1992). It means that each function is the result of the natural processes of the total ecological sub-system which it is a part of (de Groot *et al.* 2002). Accordingly, the following four classes of forest functions were then selected: (i) the capacity of forests to provide raw materials (productive function); (ii) the capacity of forests to regulate runoff and to prevent floods and soil erosion phenomena (protective function); (iii) the capacity of forests to provide habitat for wild plant and animal species (ecological-conservative function); and (iv) the capacity of forests to both filter and store fresh water, and to provide opportunities for recreational uses (other functions). During this phase, the expert also indicated a reliance between functions and a list of main classification criteria, as reported in Table 14.

Table 14: List of the classification adopted during the functions assignment process. This table reports also a brief explanation for each criterion.

Criteria	Description
Current forest management	Analysis of forest management evidences on trees (e.g. cuts, paint signs, numbers of managed forest areas, presence/absence of forest tracks, etc.).
Forest Type, forest structure and dendrometric characteristics	Analysis and evaluation of: tree species composition and forest cover; forest density; average basal area; average diameter at breast height; average and dominant tree height.
Geo-pedological conditions	Analysis of: prevalent forest site slope; altitude; presence/absence of hydrological instability phenomena; limiting factors to root expansion.
Ecological conditions	Analysis of: number of species richness (tree, shrub and herbaceous); number of micro-habitats of natural interest.

During the field surveys, such preliminary indications were used to assign a prevalent function to the investigated forest stands, in the following way: (i) recommended (R), when a correlation between the forest function and the criterion exists; (ii) not recommended (NR), when a correlation between the forest function and the criterion does not exists; (iii) irrelevant (I), when the forest function is not dependent on a specific criterion. Only in the first case (R), the technicians assessed the correlation degree by using a score-scale from 1 to 5 (from low to high correlation). Table 15 summarizes the functions' assignment process.

Table 15: Table of correlation between forest functions and classification criteria. Correlation degrees are reported in brackets. FT: Forest Type; R: Recommended; NR: Not recommended; I: Irrelevant.



	At stand level	At tree level
	Biotic and abiotic damages Presence or absence of human activities Human-made infrastructures Forest management system Presence or absence of microhabitats Forest regeneration capacity Presence or absence of mushrooms or truffles Presence or absence of lichens	
Quantitative attributes	Location Coordinates Main altitude Main slope Main aspect Tree species composition Forest cover Amount of deadwood	Diameter at breast height (DBH) Basal area Canopy height

k-NN spatialization of forest attributes

Generally, several forest stand parameters have been used at different steps within the MCML approach (see Table 17), while only three of those have been mapped by adopting the *k*-NN method and using the K-NN FOREST software (Chirici *et al.* 2012), such as: (i) the average basal area per hectare (G); (ii) the current average height (H_r); (iii) the number of tree species per hectare (TSN).

Table 17: Characteristics and brief descriptions of forest attributes which have been used throughout the mapping process. For each forest attribute, the minimum and maximum values, the standard deviation (SD), and the coefficient of variation (CV) are also reported.

Forest attribute and abbreviation	Measurement unit	Description	Min value	Max value	SD	CV
Basal area (G)	$m^2\ ha^{-1}$	Obtained by surveys data processing, as a parameter of forest productivity.	2.44	65.02	13.70	0.46
Current tree height (Hr)	m	Obtained by surveys data processing, it is used, combined with the following parameter, as site fertility index.	5.18	27.93	5.44	0.35
Potential tree height (Hn)	m	Resulting by local single-tree growth models (Castellani 1982), it has been selected for each forest type of Molise. If compared with H_r, it provides useful indications about possible forest stand productivity attitude.	4.40	25.62	4.65	0.34
Main slope (S)	degrees	Directly derived from DEM, it has been considered as a proxy of the risks linked to superficial stony-rolling or landslides and the limits to roots-growth.	0	65.35	10.55	0.56
Tree species number (TSN)	$n\ ha^{-1}$	Resulting by qualitative surveys, it represents the richness of tree species constituting the forest stand structure. It is an index of the current tree biodiversity.	1	9	1.90	0.71

A general review of the *k*-NN approach can be found in McRoberts and Tomppo (2007). A complete description of the adopted *k*-NN procedure is available in Chirici *et al.* (2008). Conceptually the unknown value of the target variable $\hat{y}_t$ for the unit (pixel or pixels group) t of the target set can be estimated using the values y_i of the same variable measured in the field in plots corresponding to the k nearest neighbours (in the multidimensional space defined by the spectral signature in the remote sensed images) units of the reference set, as reported in the equation (1).

$$\hat{y}_t = \frac{\sum_{i=1}^{k} w_{t,i}\, y_i}{\sum_{i=1}^{k} w_{t,i}} \qquad (1)$$

where: the weight $w_{t,i}$ is inversely proportional to the multidimensional distance between the units t and i measured on the n-dimensional feature space, n is the number of the feature space variables.

For the study area, the IRS-P6 image pixels – for which the forest inventory observations were available – have been denoted as the reference set in agreement with the nomenclature used by McRoberts and Tomppo (2007). 102 plots have been used as reference set for estimating both G and H_r, and 117 plots for TSN. The three above-mentioned forest attributes to be estimated for the target set have been denoted as target variables. The four original bands from the IRS-P6 image were averaged within a 3x3 pixel created around the centre of the plots of each used reference set.

Theoretically, the multidimensional distance can be calculated by several measures (e.g., in *k*-NN FOREST software three different distance measures are available; Chirici *et al.* 2012) . After an optimization phase, in this study we adopted the Distance weighted with Fuzzy weights or Fuzzy Distance (FD) for estimating H_r and TSN, and the Euclidean Distance (ED) for estimating G. For a full description of these measures, see Chirici *et al.* (2008; 2012). The optimization phase in *k*-NN FOREST software has been performed by adopting the Leave-One-Out (LOO) cross-validation procedure, the Pearson correlation index (r) and the Root Mean Square Error (RMSE) between *k*-NN estimates and the measured response variable values. These validation tools have been computed for all the reference set, according to the procedure proposed by Franco-Lopez *et al.* (2001). After the optimization procedure, a $k = 6$ was set for the three *k*-NN estimations of the pixels belonging to the target set. The final accuracy of the *k*-NN estimations is expressed by the relative percent of RMSE, calculated by dividing the RMSE for the measured mean value of the reference set (see, Fazakas *et al.* 1999).

Implementing the Multi-Criteria and Multi-Level approach

In order to map the FDUs, the forest attributes maps (G; H_r; TSN), the slope map (S) derived from DEM and the FTs map have been used as input layers in the proposed MCML approach. The approach is defined multi-criteria because it uses exclusion selection criteria (or…or) and is based on the restricted selection of chosen attributes of interests (G, H_r, S or TSN); and multi-level because the process phases develop on four hierarchical levels. The progression is reached only if the previous criteria are respected. A summary description of the MCML approach is given in Table 18, and a related flowchart is shown in Figure 15.

Table 18: Related contents to the MCML approach: relationships among the discriminating criteria for each level.

Level one	Discriminating criteria: Slope. The 1st level regards the DEM Mask overlapping in order to evaluate the slope conditions for each pixel: - If Slope < 75%, then it jumps to level two, following the Branch A; - If Slope > 75%, then it passes to Branch B and assigns the pixel to PROTECTIVE FDU directly.
Level two	Discriminating criteria: G. The 2nd level applies the G Mask overlapping in order to evaluate the G values for each pixel: - If G > 30 m^2/ha, then it passes to Branch C and assigns the pixel to PRODUCTIVE FDU directly; - If G < 30 m^2/ha, then it jumps to Level three, following the Branch D.
Level three	Discriminating criteria: H_r. The 3rd level uses the Hr Mask overlapping in order to evaluate the H_r for each pixel: - If H_r>H_N, then it passes to Branch E and assigns the pixel to PRODUCTIVE FDU directly; - If H_r<H_N, then it jumps to level four, following the Branch F.
Level four	Discriminating criteria: TSN. The 4th level uses the informative layer TSN, and it associates each pixels to a specific number of tree species in the investigated forest stands. Specifically: - If N species < 3, then it passes to Branch G and assigns the pixel to OTHER FDUs; - If N species > 3, then it passes to Branch H and assigns the pixel to ECOLOGICAL-CONSERVATIVE FDU.

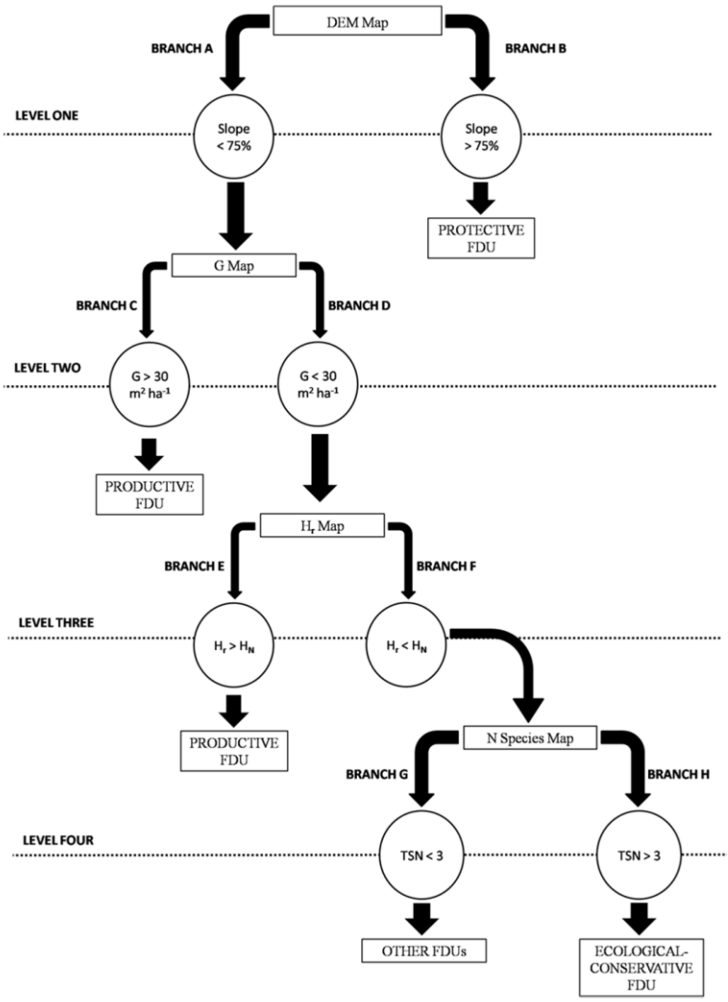


Figure 15: The MCML approach.

Agreement assessment

The agreement degree between pixels whose forest function was classified in the field survey phase and the pixels geographically correspondent in the resulting FDUs map was also calculated. The concordant pixels are reported in the diagonal of the resulting error matrix. Outside the diagonal the matrix reports the discordance between pixels. The errors (discordance in the classification) are divided into Commission Errors (CE; pixels refer to a specific class, but they have not been classified for that) and Omission Errors (OE; points wrongly classified for a given class). The ratio among the number of points on the diagonal and the total points of the correspondent row represents the Producer Accuracy (PA), while the ratio between the number of points on the diagonal and the total points of the row represents the User Accuracy (UA; Corona 1999). These evaluations are available for each class. In addition, the Overall Agreement (OA) represents the Percentage of concordance between classified pixels (PCC). The PCC pixels is expressed by the formula (2).

$$PCC = \frac{\sum_{j=1}^{C} n_j}{n} \qquad (2)$$

where: n_j is number of sampling points that have been correctly attributed to the *j*-th thematic class; *n* is the total number of points; and *C* is the number of thematic classes.

3.3.3 Results

Figure 16 reports the distribution of the three main forest attributes (Hr, G, and TSN) per forest function which were then spatialized adopting the *k*-NN method.

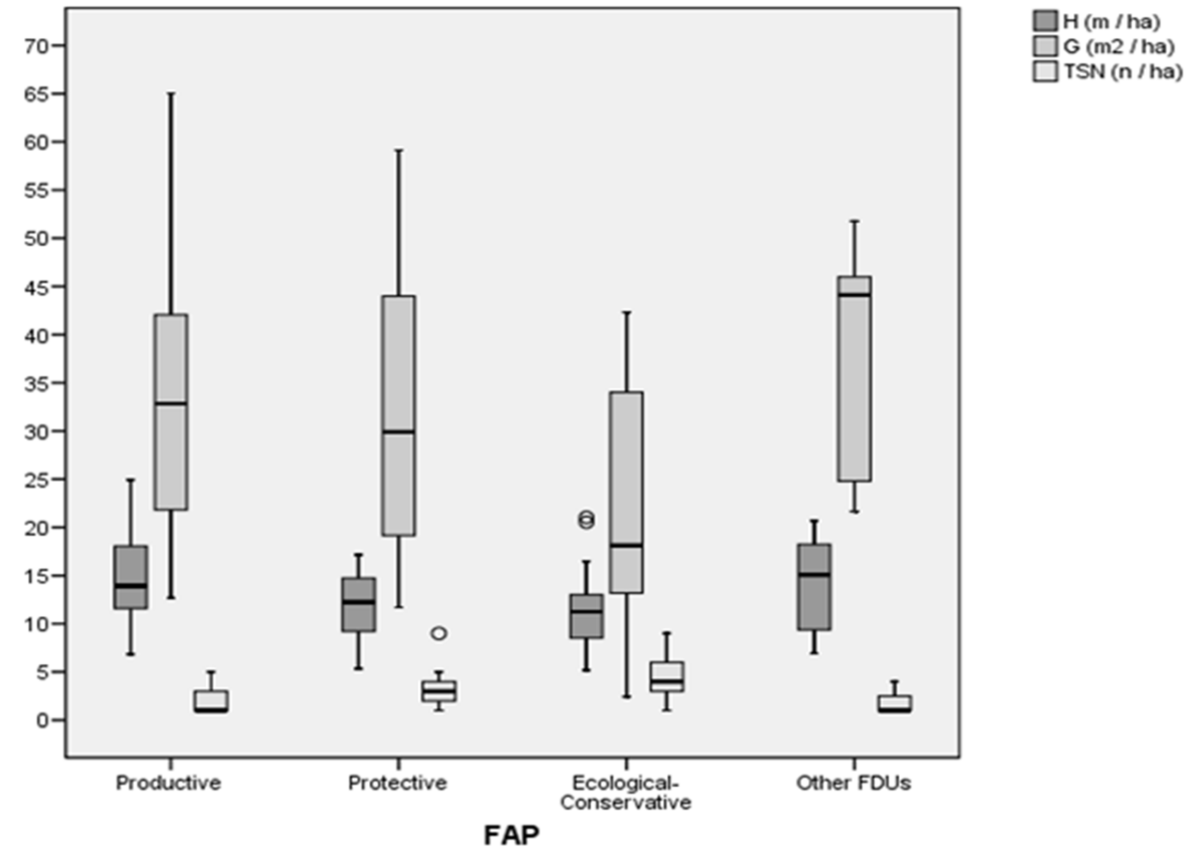


Figure 16: Box-plot showing the distribution of the collected forest attributes between the assigned FDUs.

Analyzing the forest attributes distribution, the following insights can be described: (i) in H_r distribution, the inter-quartile range (IQR) value is the highest for the other FDUs (about 9 m ha^{-1}); (ii) H_r distribution is negatively skewed for the productive function, and positively skewed for the other FDUs; (iii) G distribution is negatively skewed for the ecological-conservative function, and positively skewed for the other FDUs; (iv) in G distribution, IQR value is the highest for the protective function (more than 24 m^2 ha^{-1}); (v) TSN distribution is negatively skewed for productive and ecological-conservative functions, and for other FDUs; (vi) in TSN distribution, IQR value is the highest for the ecological-conservative function (3 n ha^{-1}); (vii) 3 outliers have been found (1 concerning TSN for the protective function, and 2 concerning H_r for the ecological-conservative function).

The estimated forest attributes maps are shown in Figure 17(a-c).

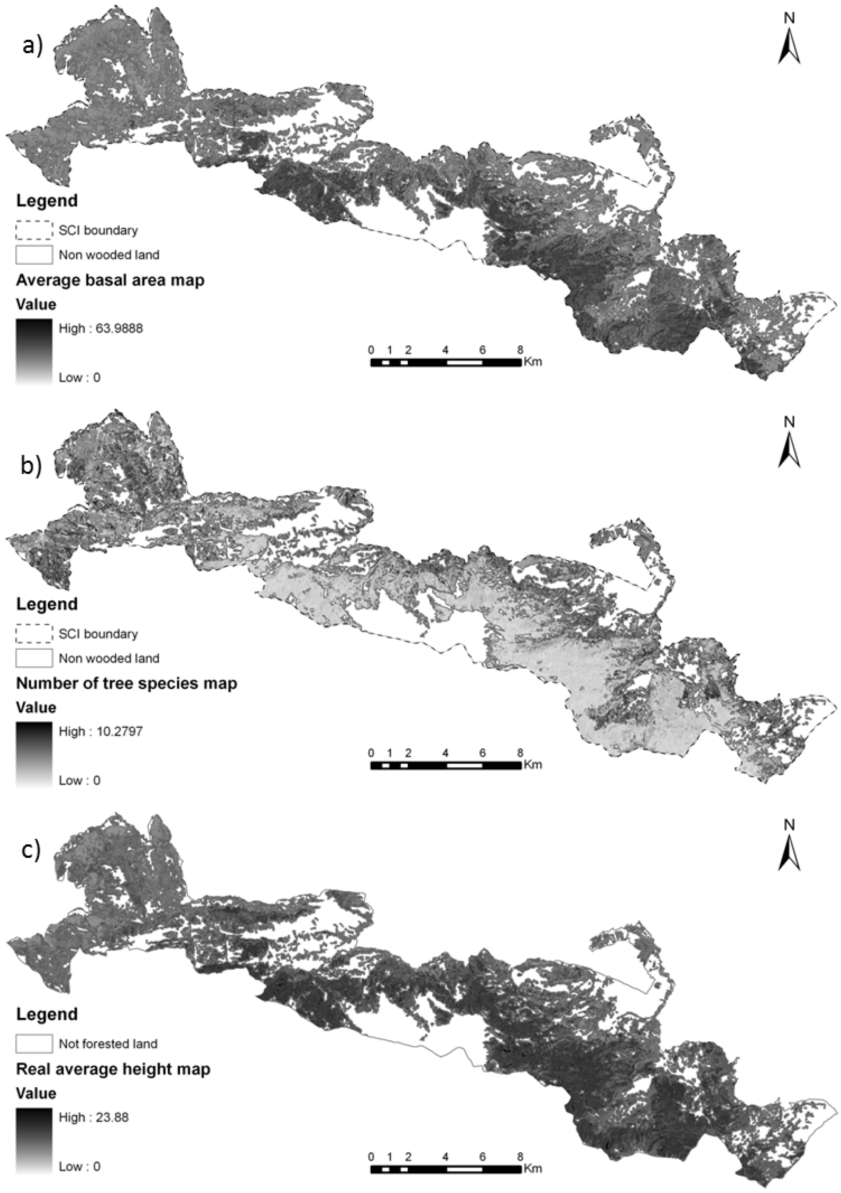


Figure 17: G, TSN and H_r maps as resulted by the *k*-NN spatialisation process (a, b, and c, respectively).

In particular, Figure 17(a) shows that G increases at the highest altitudes and in steep conditions, while decreases in lowland conditions, or in correspondences of creeks, valleys and ravines. This is mainly due to the absence of past human interventions (silvicultural practices and harvesting operations) and infrastructures (forest paths and roads) at both the higher altitudes and in steeper conditions. Adversely, TSN is higher in lowlands than on the mountain peaks (see Figure 17(b)). No important differences about the Hr distribution are denoted (see Figure 17(c)). The final accuracy of the pixel level G estimation (expressed here as the relative percent RMSE) is 0.8%. G as measured in the reference set was 31.4 m^2 ha^{-1}, ranging from 2.4 to 65 m^2 ha^{-1}. G as estimated with *k*-NN in the target set was 30.9 m^2 ha^{-1}, ranging from zero to 63.9 m^2 ha^{-1}. The relative percent RMSE of the pixel level H_r estimation is 2.5%. H_r as measured in the reference set was 13.5 m ha^{-1}, ranging from 5.2 to 24.9 m ha^{-1}. H_r

as estimated with *k*-NN in the target set was 13.3 m ha^{-1}, ranging from zero to 23.6 m ha^{-1}. TSN as measured in the reference set was 4, ranging from zero to 11 species. TSN as estimated with *k*-NN in the target set was equal to that measured (4 species), ranging from zero to 9 species. The final accuracy expressed as relative percent RMSE is 2.5%.

The final result of the MCML process is the FDUs map (Figure 18).

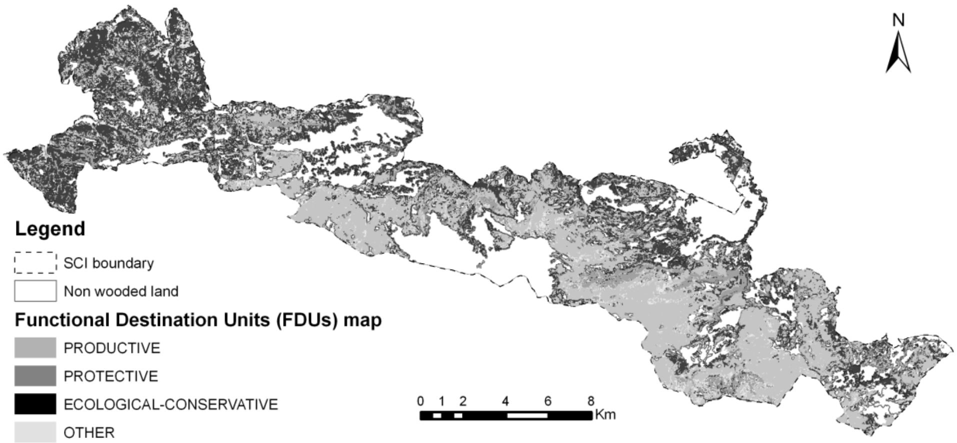


Figure 18: FDUs map.

The FDUs map is a thematic raster product showing all prevalent forest functions associated to all forest area pixels. This means that, randomly choosing a point on the map, it is possible to know its primary forest ecosystem function. With regard to forested area within the Matese landscape, the FDUs map shows that FDUs are distributed as follows: 48% productive, 7% protective, 38% ecological-conservative, and 8% other FDUs. Results of the FDUs map classification accuracy are reported in Table 19.

Table 19: Error matrix. The numbers represent classified plots. Columns represent plots classified in the field survey phase. Rows represents plots classified through the MCML approach.

		Pixels related to functions assigned					
		Productive	Protective	Ecological-conservative	Other	Tot. Row	UA
Fdusmapclassespixels	Productive	41	8	12	0	61	67%
	Protective	0	3	1	2	6	50%
	Ecological-conservative	3	3	25	0	31	81%
	Other	8	4	5	2	19	10%
	Tot. Column	52	18	43	4	117	
	PA	79%	17%	58%	50%		
				OA		61%	

Table 20 shows a synthesis report of the agreement assessment.

Table 20: Summary table of the main results obtained by error matrix.

FDUs Classes	PA	UA	OA
Productive	79%	67%	61%
Protective	17%	50%	
Ecological-conservative	58%	81%	
Other FDUs	50%	10%	

FDUs map results in good values concerning the overall classification accuracy (OA=61%). Moreover, a very good value of user accuracy (UA=81%) for ecological-conservative FDU shows a correct assignment of pixels by FDUs map to this class. Another excellent result has been obtained in producer accuracy (PA=79%) for productive FDU, demonstrating a correct assignment of related-pixels by producer to this FDU.

3.3.4 Discussion and conclusions

The FDUs distribution is strictly linked to the explored landscape. It expresses characteristics and potentialities of investigated forest resources, according to their prevalent forest function. The proposed research methodology is reliable to identify and map those forest units that are able to provide a prevalent forest function – named here FDU – to be enhanced in the context of FLMP-related decision-making processes.

The sampling plots distribution among forest attributes generally demonstrates a strong correlation between the functions' assignment and the current conditions of the investigated forest stands. H_r and G values tend to decrease when passing from productive to protective, and to ecological-conservative FDUs, thus demonstrating that forest attributes relating to forest productivity and site fertility (such as H_r and G) were correctly associated with the productive FDU. The TSN values increase when passing from productive to ecological-conservative FDU. Higher H_r and G values are related to the other FDUs. During the field surveys phase, only seven sampling plots were correlated to the other FDUs, which had similar characteristics, such as: (i) presence of big-dimension trees and old-growthness conditions; (ii) low forest stand density; (iii) absence of recent forest management practices or silvicultural interventions; and (iv) diffuse presence of tourism-related infrastructures and dedicated areas for recreational purposes. More generally, H_r, G and TSN distributions are skewed, excepting for G distribution in the cases of both productive and protective FDUs. In addition, the highest IQR values resulted for: (i) H_r distribution in the case of the other FDUs; (ii) G distribution for the protective FDU; (iii) TSN distribution in the case of the ecological-conservative FDU. These results may depend on an overall variability among the investigated forest stands and by the plots' frequency for each assigned FDU.

Considering the accuracy of the *k*-NN spatialisation of the forest attributes, our relative RMSE values are significantly lower than those obtained in other previous works, despite their different aims (for G, see, e.g., Tuominen and Pekkarinen 2004 and Packalén and Maltamo 2007; for H_r and TSN, see Holmström 2002; and for G and Hr, see Järnstedt *et al.* 2012).

Final results demonstrated that mapping FDUs by adopting the proposed MCML approach, the use of inventory data and their combination with remote-sensed images constitute a feasible integrated approach. The FDUs map is the final product of an applied methodological scheme and it could be helpful to support decision-making processes into the FLMP of the Matese forests. Since FLMP provides the forest management guide-lines to implement the SFM principles at landscape level according to the balance between forest resources, rural framework and the local inhabitants needs, our methodology is proved to support the FLMP implementation. In previous experiences (see, e.g., Cantiani *et al.* 2010, Paletto *et al.* 2012, and Di Salvatore *et al.* 2013), the forest functions have been assigned by expert knowledge with no mapping approach. Thus, the integration of inventory and mapping approaches should be further developed in order to support the assessment of forest ecosystem functions, especially at landscape level. Furthermore, FDUs map can be used as an important tool to support the decision-making processes according to a forest functions framework within forested landscapes. Indeed, the FDUs map can be considered as the starting point to develop the forest management planning at larger scales and thus be helpful to drive the stakeholders into structuring the SFM Indicators Networks (Santopuoli *et al.* 2012). For example, linking the FDUs map with a forest management plan can orient the forest management planners towards alternative schemes of forest interventions , in agreement with the current forests' conditions and with regards to the main ecosystem functions they can actually provide. In addition, the proposed MCML approach can be used to further structure many decision-making processes at landscape level not only in Italy, with considerations to the same forest stands' conditions, e.g. along mountains forest areas.

Evaluating and mapping forest ecosystems functions are some of the key issues in forest management planning at landscape level (Cantiani *et al.* 2009). This paper showed an objective and cost- and time-efficient methodology to analyze and quantify the current state of main forest ecosystem functions. The GIS-based approach presented in this paper has been conceived to provide a technical support for the sub-regional environmental zoning and ecosystem-based management planning. In many cases, a visual representation of fragmented forest areas, and of emerging issues regarding SFM, biodiversity, and ecosystem-based management assumes an intellectual and practical significances (Chen *et al.* 2009). Forest planning managers would be interested in visually displaying the extent of ecosystem functions by some geographical units with management significance, such as town, county, or watershed (Troy and Wilson 2006).

Mapping ecosystem functions (and the consequent services flow) at higher levels of spatial accuracy is crucial to assist decision makers in identifying priority areas for management and conservation of natural resources (see also Troy and Wilson 2006, and Naidoo *et al.* 2008). Especially in the case of forest ecosystems, SFM depends on the participation of a wide range of stakeholders (Hamersley-Chambers and Beckley 2003; Raison *et al.* 2001). Accordingly, mapping FDUs (in a broader sense, forest ecosystem functions) can lead forest management planners, local stakeholders and communities towards a better understanding of the primary ecosystem functions provided by the forest areas, as well as towards a multipurpose forest management at landscape level. Moreover, the main outcomes of this paper provide a first decision support tool to automatically describe and analyze the current state of forest resources by assessing their primary functions, thus supporting the forest management planning decisions at landscape scale. In fact, FLMP was originally conceived as a knowledge-based forest management tool, in order to address specific management guidelines towards a sustainable way (Agnoloni *et al.* 2009). This can be achieved only evaluating the productive, protective and ecological-conservative characteristics of the investigated forest areas, as well as their spatial distribution. Furthermore, assessing forest ecosystem functions in a Natura2000 Network site is of primary importance to improve and enhance biological diversity conservation, thus avoiding an overexploitation of natural resources and considering their regeneration capacity. Indeed, this tool can be tuned to allow a spatially optimised allocation of economic resources to preserve many important forest ecosystems properties, such as the capacities to provide non-timber products, to protect human settlements from natural risks, or to preserve the cultural and spiritual heritages. Thus, this methodological approach can be widely used in further research activities in both Natura2000 Network or non-protected areas.

At conclusion, the following considerations have to be pointed out: (i) despite the proposed methodology can be considered as experimental in linking forest inventory and mapping approaches, it can be useful for forest planners and practitioners who are facing with decision-making problems in forest management planning; (ii) our methodological approach can be easily replicated and adapted to other forest landscapes in Italy; (iii) the spatial estimation of forest attributes proved to be a cheap and feasible intermediate tool to map forest ecosystem functions, thus improving the economic assessment and promoting the sustainable forest management at a landscape level.

3.4 Case study 3: Assessing barriers and drivers for Integrated Forest Management at landscape scale in Italy[6]

3.4.1 The context

Forest ecosystem services (FES) are crucial for sustaining local economies and for enhancing the well-being of populations living in rural and marginal areas (Sunderlin *et al.* 2008; Persha *et al.* 2011). Over the past, forest management (FM) has evolved with the primary purpose to maximize the economic income from timber (Puettmann 2011). More recently, forest managers have been called to balance timber production with alternative services (e.g. carbon sequestration, recreation and tourism, non-timber forest products, etc.) in agreement with the approaches of sustainability (Wilkie *et al.* 2003) and adaptation (Holling 1978; Walters 1986). In addition, several studies on the ES assessment asked for: (i) alternative tools to assess the ES potential (Busch *et al.* 2012) in order to bridge the gap between sectorial management landscape approaches and regional development planning (Frank *et al.* 2012); (ii) a better understanding of complex ecosystem functionalities or even predictions of ecosystem behavior, for example based on ecological modeling, ecological indicators, and ES (Jørgensen and Nielsen 2012); (iii) the integration of the ecosystem behaviors with plausible future-oriented scenarios about the sustainable use of ecosystem services (ES), as a basis for adaptive management (Nedkov and Burkhard 2012; HainesYoung *et al.* 2012).

In response to these emerging issues, tools originating in forest ecology (i.e. forest ecosystem models, FEM) and in operational research fields (in the sense of MCDA) have been implemented to support modern FM (Wolfslehner and Seidl 2010) through the DSS platform (see e.g. Reynolds 2005 and Reynolds *et al.* 2008). FEM are expected to allow insights into the relation of FM objectives to ecosystem dynamics, facilitate the exploration of FM options and their consequences (''what-if…'') as well as provide information on the sensitivity of systems to actions and external drivers such as climatic changes (Landsberg 2003). For an overview of FEM, the reader is referred to e.g. Porté and Bartelink (2002) or Pretzsch *et al.* (2008). By other hand, MCDA has been described as a highly feasible tool for integrated, holistic FM by providing a formal framework for participation and decision-making (see e.g. Mendoza and Prabhu 2003). An exhaustive review on forest DSS, the reader is referred to Borges *et al.* (2014).

Although the number of studies focusing on the ES assessment rapidly increased in last two decades (see e.g. Seppelt *et al.* 2011), very few publications concerning the correlation between FM and ES supply (see e.g. Duncker *et al.* 2012; Grêt-Regamey *et al.* 2013) is available. The remaining challenges for forest managers and modelers when facing the ES approach are hereinafter described. At first, stakeholders with heterogeneous backgrounds have to be included in decision-making processes, in order to utilize informal knowledge, as well as to increase legitimacy and the acceptance of decisions (Muys *et al.* 2010). Secondly, modelling ES supply as a function of processes and management interventions requires multi-

[6] Sources: Vizzarri *et al.* (2014b; 2014c; 2014d).

scale approaches that are able to incorporate drivers that function across a range of scales (de Groot *et al.* 2010; Seidl *et al.* 2013). Finally, the integration of various techniques, models and methods in a holistic and flexible manner is demanded for the new DSS developers (Vacik and Lexer 2013). Therefore, incorporating management variables into a decision model that reflect management options across these scales is important in defining 'optimal' ecosystem management approaches, but highly complex in terms of modeling (e.g. Levin 1992; Limburg *et al.* 2002).

As also stated by the European Landscape Convention (ELC), 'landscape' is a "formal expression of the numerous relationships existing in a given period between the individual or a society and a topographically defined territory, the appearance of which is the result of the action, over time, of natural and human factors and of a combination of both" (Council of Europe 2000). As a consequence, landscape conservation and management require a multidisciplinary approach, which involves policy actions reflecting all the cultural, historical, archaeological, ethnological, ecological, aesthetic, economic and social interests of the territory concerned. At landscape level, the main challenge is how to decide on the optimal allocation and management of the many different land use options (de Groot at al. 2010), so that landscape functions (and services) become an important concept in policy-making. In this context, forest ecosystems are key components of rural landscapes in Mediterranean Countries (especially in mountain environments), as well as fundamental sources of benefits for local communities. Managing forest resources for multiple purposes requires a whole understanding of dynamics and interactions in- and between ecosystems at landscape scale. Therefore, FM at landscape scale is called to balance ecosystem resilience and the demands for alternative services. In this way, the Integrated Forest Management (IFM) is a suitable approach to consider at the same time the ecological, economic and social aspects, through e.g. the enhancement of public participation in decision-making processes, or the analysis of economic barriers and drivers for sustainable development at landscape scale (see also Chapter 2).

3.4.2 Objectives and methodology

This study focuses on the implementation of a large-scale scenario model in different case studies in Italy aiming at: (i) simulate the delivery of the main FES (timber provision, carbon sequestration, biodiversity conservation, tourism and recreation) over the time, according to different scenarios; and (ii) assess in which way FES provision is influenced by several driving forces at landscape level, such as management drivers, ecological components, and landscape features, in the frame of IFM. The methodology is structured into the following steps: (i) selection of case studies and collection of forest attributes; (ii) building of alternative future-oriented scenarios; (iii) FES assessment; (iv) assessment of factors impacting the FES provision; and (v) EFISCEN model parameterization and running.

Selection of case studies and collection of forest attributes

For this study, the following sites were selected as representative forest landscapes in Italy: (i) the Asiago Municipality forest area, Veneto region, Northern Italy; (ii) the Collemeluccio-Montedimezzo UNESCO Man and Biosphere Reserve (MaB), Molise region, Central Italy; and (iii) the North-western area of Etna mountain (part of the Natural Park of Etna mountain). Table 21 reports the most important characteristics of selected sites.

Table 21: Main characteristics of selected forest landscapes.

Site name (and acronym)	Landscape description
Asiago Municipality forest area (ASI)	The Asiago Municipality is located in the North-central part of Veneto region, Northern Italy. The forest area is about 10,300 ha, of which 5,900 ha are managed. The altitude range shifts from 199 m a.s.l. to 2,310 m a.s.l. The annual average precipitations are comprised between 1,500 and 1,800 mm year-1. The annual average temperature is 7°C. The total forest management area is divided into 238 forest management units (grouped into 5 forest compartments), which are structured into the following categories: productive high-forests (46.6%); protective forests and shrublands (17.9%); European beech coppice forests (10.9%); Natural Reserve (9.5%); pastures (15.2%). The main forest categories are following reported as percentages on the total forest area: beech forests (27.6%); Norway spruce forests (26.2%); silver fir forests (15.4%); coniferous plantations (12.7%).
Collemeluccio-Montedimezzo UNESCO MaB (COM)	The 'Collemeluccio-Montedimezzo' UNESCO Man and Biosphere (MaB) Reserve is located in Molise region, Central Italy, and covers an area of 637 ha, entirely forested. The Reserve is divided into two sub-areas, such as Montedimezzo and Collemeluccio. The landscape is characterized by a sub-mountainous range (elevation from 800 to 1,277 m a.s.l.) with a various pattern of reliefs and fluvial plains. In Montedimezzo, the annual precipitation is 1,012 mm and the annual average temperature is 8.5°C. In Collemeluccio, the annual precipitation is 916 mm and the annual average temperature is 8.4°C. In Montedimezzo, forests are mainly composed by Turkey oak (50%) and beech (50%) stands. In Collemeluccio, forests are mainly composed by silver fir (70%) and Turkey oak (30%) stands. The silver fir population is a relic of the last Ice Age and has a great historical and phyto-geographical value as well as being an important genetic resource due to its differentiated population. Forest ownership is entirely public.
North-western area of Etna mountain (ETN)	The North-western area of Etna mountain is located in Sicilian region, Southern Italy, and covers an area of 25,225 ha. The area includes three municipalities, such as Bronte, Maletto, and Randazzo. The altitude range shifts from 375 m a.s.l. to 3,300 m a.s.l. Rainfalls are about 651.73 mm year-1. The average annual temperature is 17°C. Forests cover about 7,000 ha (27% of the total area). The main forest categories are hereinafter reported as percentages on the total forest area: pure and mixed deciduous oak forests (mainly dominated by downy oak) (23%); mountainous shrublands (17%); holm oak forests (16%); and beech forests (10%). Considering the natural areas, the North-western area of Etna mountain is mainly comprised within the Etna Regional Park (about 19,910 ha). Forest ownership is mainly private with small and fragmented lots (about 80% of the total area).

For each case-study a forest management plan (FMP) was available. Each FMP reports the parameters for each forest parcel/forest management unit (i.e. forest stand), as follows: (i) average standing volume (m^3 ha^{-1}); (ii) forest management system applied (coppice forest, even-aged high forest, uneven-aged high forest, high-coppice forest); (iii) average current annual increment (m^3 ha^{-1} $year^{-1}$); tree species composition (tree species or groups of species); (iv) additional parameters, such as: average basal area per hectare (G, m^2 ha^{-1}), average diameter per hectare (dg, cm), and dominant tree height (dH, m).

Future-oriented scenarios building

Future-oriented scenarios of IFM were built through the consultation of local stakeholders. Several thematic workshops and meetings were organized in each case study in order to analyse the main factors influencing IFM at landscape level, and thus coherently building case-specific alternative future-oriented scenarios (namely, 'driver' scenarios). Accordingly, 'driver' scenarios represent different future configurations of investigated landscapes, which develop on the basis of the interactions between structural factors (e.g. demographic structure, social developments, policy coherence, technological changes, etc.) and agent-based factors (e.g. local stakeholders interests/preferences, decision-making authority, actions and strategies, cooperation, etc.). Table 22 reports the details about scenarios for each case-study area.

Table 22: Main characteristics of scenarios obtained by the participatory process. [1]Excepting than for the scenario 1, the other scenarios can be grouped in three intensity and frequency levels of forest management over the time, as follows: (i) 'integral', in which the forest management is based on 'close-to-nature' and 'holistic' approaches, and biodiversity conservation, natural regeneration and habitat stability are respected (scenarios 3 and 7); (ii) 'more active', in which forest management is oriented to improve the timber production and the economic incomes (scenarios 4 and 5); and (iii) 'freezing', in which the forest management is generally avoided or totally absent (scenarios 2 and 6).

Case study	Scenario	Acronym
ASI	Worst case	ASI.SCN1
	Industrial roundwood-oriented sylviculture, bioenergy sector development at local level, Payments for Environmental Services (PESs)	ASI.SCN2
	"Bioenergization" of forests, PESs development	ASI.SCN3
	High-quality trees sylviculture, bioenergy sector development at local level, PESs	ASI.SCN4
	Business as usual (BaU) prevailing conditions	ASI.SCN5
COM	Worst case	COM.SCN1
	MaB suppression, bioenergy sector development, unstructured tourism development	COM.SCN2
	Best case	COM.SCN3
	Prevailing *status quo*, with integrated tourism development	COM.SCN4
ETN[1]	Worst case	ETN.SCN1
	Sylviculture "freezing" the landscape variability; PESs and integrated tourism development	ETN.SCN2
	Integral conservation and wilderness, Payment for Environmental Services (PESs) and integrated tourism development	ETN.SCN3
	More active management in B/C/D Park areas, PESs and unstructured tourism development	ETN.SCN4
	More active management in B/C/D Park areas, PESs and integrated tourism development	ETN.SCN5
	Sylviculture "freezing" the landscape variability, PESs and unstructured tourism development	ETN.SCN6
	Integral conservation and wilderness, PESs and unstructured tourism development	ETN.SCN7

Forest Ecosystem Services assessment

The following four forest ecosystem services (FES) were simulated: (i) amount of deadwood (m^3 ha^{-1} $year^{-1}$), which was considered a proxy for biodiversity conservation (see e.g. Barbati *et al.* 2014; Stokland *et al.* 2012), as supporting FES; (ii) amount of timber harvested, as provisioning FES (m^3 ha^{-1} $year^{-1}$); (iii) amount of carbon sequestered in above-ground biomass (Mg C ha^{-1} $year^{-1}$), as regulating FES; and (iv) Recreational Scores (RS; DIM), which are associated to the Forest Type, the developmental stage of the stand, and the implemented Forest Management Approach (*sensu* Edwards *et al.* 2012), as proxy for cultural and recreational FES. FES provision was assessed at the end of simulation period (30 years from now).

Assessment of factors impacting the ecosystem services provision

The differences in FES provision between scenarios are assessed by analyzing the main factors impacting the FES trend and supply per each scenario. At first, the main factors influencing the forested landscape development along the scenario's timespan were identified and classified. Then, such factors were compared to each FES as previously assessed (at the end of simulation period; i.e. 30 years). Finally, the influences (or impacts) of such factors on the FES supply were interpreted and described. In particular, FES provision was compared with impact factors (IF) by using a linear regression, and according to equation (3):

$$FES = a \pm b(IF) \qquad (3)$$

where: FES is the considered Forest Ecosystem Service; a is the intercept of the regression line; $\pm b$ represents the positive/negative impact (slope of regression line); and IF is the considered impact factor.

Although several factors currently influence the forested landscape development over the time (e.g. intensity and frequency of harvesting operations, amount of removals released, age distribution, tree species composition, length of rotation period, land ownership structure, local climate, soil fertility, etc.), only four factors were taken into account, such as: (i) intensity of interventions; (ii) stand ages distribution; (iii) tree species composition; and (iv) ownership structure. Table 23 summarizes the main factors impacting the FES provision for all scenarios in each case study. The impact of factors on FES provision was assessed at the end of simulation period (30 years from now).

Table 23: List of main factors impacting the FES provision in case studies.

Factor Class	Factor Type	Acronym	Description	Measurement unit
Management driver	Intensity of interventions	VH	Ratio between the volume harvested and the growing stock (GS) of volume available	DIM [%GS]
Ecological component	Stand ages distribution (SAD)	SAD	Forest area whose age is more than 150 years (ageing stands)	ha
Ecological component	Tree species composition (TSC)	SC	Forest area covered by broadleaved species (mixed stands)	ha
Landscape feature	Ownership structure (FOT)	FOT	Area sharing among different owner types	DIM [% total area]

EFISCEN model parameterization and running

In this study, the European Forest Information SCENario (EFISCEN) model (Nabuurs 2001, Schelhaas *et al.* 2007) was used. EFISCEN was originally developed by Sallnäs (1990) as an area-based matrix model. For each forest type (or forest category) that is distinguished in the input data (according to species, region, site class and owner), a separate matrix is set up. Generally, one matrix consists of 60 age classes of 5-year width and 10 volume classes. Ageing of forest is simulated by moving the area to a higher age class, while growth is simulated by moving the area to a higher volume class. Thinning in the model is simulated by moving the area one volume class down. Final fellings are simulated by taking the area out of a certain cell of the matrix. Natural mortality is simulated by moving a fraction of the area in a certain cell one volume class down.

Table 24 reports the main forest information that were used as input parameters for EFISCEN model in all three case studies.

Table 24: List of input parameters.

Parameter	Description	Source
Forest Type	8 Forest Types: ASI: (i) subalpine and mountainous spruce and spruce-silver fir mixed forest (even- and uneven-aged high forests); and (ii) Illyrian mountainous beech forest (even-aged, uneven-aged and coppice forests). COM: (i) Apennine-Corsican mountainous beech forest (even- and uneven-aged high forests, and coppice forests); and (ii) Turkey oak, Hungarian oak and Sessile oak forest (even- and uneven-aged high forests, and coppice forests); and (iii) Mediterranean and Anatolian fir forest (even- and uneven-aged high forests). ETN: (i) Apennine-Corsican mountainous beech forest; (ii) holm oak and downy oak forests (mainly coppice forests); (iii) Mediterranean and Anatolian black pine forests (even- and uneven-aged high forests)	EEA (2006)
Age class	10-year step	Local Forest Management Plan
Site index	3 soil fertility index	
Forest area	ha	
GS	$m^3\ ha^{-1}$	

Parameter	Description	Source
Current Annual Increment	m^3 ha^{-1} $year^{-1}$	
Forest Management System	Intensity and Frequency of interventions	
Climate data	DIM	Information at national scale

3.4.3 Results

Forest Ecosystem Services provision trade-offs among scenarios

Figure 19 reports the trade-offs about FES provision between scenarios per each case study.

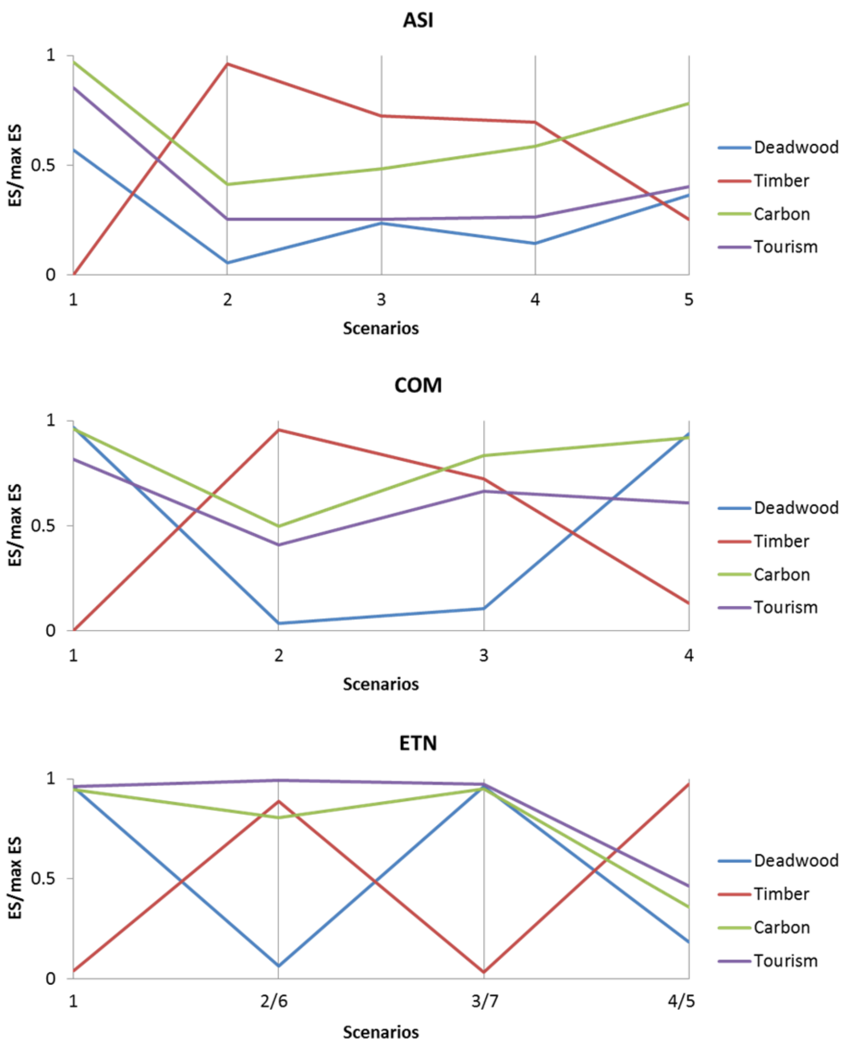


Figure 19: FES trade-offs between scenarios per each case study at the end of simulation period (30 years from now). FES values are reported in terms of the ratio between the FES value of a given scenario and the maximum FES value for all considered scenarios (ES/ max ES).

Details about FES values for ASI case study are hereinafter reported. The total amount of deadwood resulted as follows: (i) 11,911 m^3 (scenario 1); (ii) 1,443 m^3 (scenario 2); (iii) 5,070 m^3 (scenario 3); (iv) 1,984 m^3 (scenario 4); and (v) 7,429 m^3 (scenario 5). The amount of deadwood increases in Scenario 1 (86 m^3 $year^{-1}$), Scenario 2 (33 m^3 $year^{-1}$), Scenario 3 (22 m^3 $year^{-1}$) and Scenario 5 (80 m^3 $year^{-1}$). The amount of deadwood decreases in Scenario 4 (76 m^3 $year^{-1}$). The total amount of timber harvested resulted as follows: (i) 0 m^3 (scenario 1); (ii) 102,121 m^3 (scenario 2); (iii) 33,438 (scenario 3); (iv) 115,175 m^3 (scenario 4); and (v) 29,206 m^3 (scenario 5). The amount of timber harvested increases more in Scenario 2 (3,404 m^3 $year^{-1}$) and Scenario 3 (1,115 m^3 $year^{-1}$), and in Scenario 4 (3,839 m^3 $year^{-1}$) than in Scenario 5 (974 m^3 $year^{-1}$). In Scenario 1, the amount of timber harvested is about 0 m^3. The total amount of carbon sequestered in above-ground biomass resulted as follows: (i) 705,806 Mg (scenario 1); (ii) 290,098 Mg (scenario 2); (iii) 345,737 Mg (scenario 3); (iv) 378,947 Mg (scenario 4); and (v) 560,054 Mg (scenario 5). The amount of carbon sequestered increases in Scenario 1 (7,352 Mg $year^{-1}$) and Scenario 5 (103 Mg $year^{-1}$). The amount of carbon sequestered strongly decreases in Scenario 2 (8,895 Mg $year^{-1}$), Scenario 3 (7,040 Mg $year^{-1}$), and Scenario 4 (5,933 Mg $year^{-1}$). The total RS resulted as follows: (i) 0.029 (scenario 1); (ii) 0.010 (scenarios 2 and 3); (iii) 0.009 (scenario 4); and (iv) 0.016 (scenario 5). The RS decrease in Scenario 2 (3.6), Scenario 3 (3.6), Scenario 4 (14.8), and Scenario 5 (2.2). The RS increases only in Scenario 1 (0.5%).

Details about FES values for COM case study are hereinafter reported. The total amount of deadwood resulted as follows: (i) 6,208 m^3 (scenario 1); (ii) 339 m^3 (scenario 2); (iii) 416 m^3 (scenario 3); and (iv) 6,040 m^3 (scenario 4). The amount of deadwood increases in Scenario 1 (about 56 m^3 $year^{-1}$), Scenario 2 (11.3 m^3 $year^{-1}$), Scenario 3 (about 9 m^3 $year^{-1}$), and Scenario 4 (51.3 m^3 $year^{-1}$). The total amount of timber harvested resulted as follows: (i) 0 m^3 (scenario 1); (ii) 14,548 m^3 (scenario 2); (iii) 26,711 m^3 (scenario 3); and (iv) 2,212 m^3 (scenario 4). The amount of timber harvested is proximal to 0 in the case of Scenario 1, while it increases in Scenario 2 (485 m^3 $year^{-1}$), Scenario 3 (890 m^3 $year^{-1}$), and Scenario 4 (74 m^3 $year^{-1}$). The total amount of carbon sequestered in above-ground biomass resulted as follows: (i) 216,062 Mg (scenario 1); (ii) 126,065 Mg (scenario 2); (iii) 189,673 Mg (scenario 3); and (iv) 207,799 Mg (scenario 4). Carbon sequestration increases in Scenario 1 (2,176 Mg $year^{-1}$), Scenario 3 (about 1,300 Mg $year^{-1}$), and Scenario 4 (1,900 Mg $year^{-1}$). It decreases in Scenario 2 (824 Mg $year^{-1}$). The total RS resulted as follows: (i) 0.030 (scenario 1); (ii) 0.018 (scenarios 2); (iii) 0.025 (scenario 3); and (iv) 0.045 (scenario 4). RS increase in Scenario 1 (9.4), Scenario 3 (about 12), and Scenario 4 (about 9). They decrease in Scenario 2 (0.9).

Details about FES values for ETN case study are hereinafter reported. The total amount of deadwood resulted as follows: (i) 36,774 m^3 (scenario 1); (ii) 1,434 m^3 ('freezing' scenarios); (iii) 36,855 m^3 ('integral' scenarios); and (iv) 6,667 m^3 ('more active' scenarios). The amount of deadwood increases in Scenario 1 (about 830 m^3 $year^{-1}$), and Scenarios 3 and 7 (833 m^3 $year^{-1}$), while it decreases in Scenarios 2 and 6 (about 23 m^3 $year^{-1}$), and 4 and 5 (0.34 m^3 $year^{-1}$). The total amount of timber harvested resulted as follows: (i) 18,948 m^3 (scenario 1); (ii) 271,284 m^3 ('freezing' scenarios); (iii) 11,929 m^3 ('integral' scenarios); and (iv) 792,686 m^3 ('more active' scenarios). The timber harvested increases in Scenario 1 (0.63 m^3 $year^{-1}$), Scenarios 2 and 6

(9.04 m^3 $year^{-1}$), Scenarios 3 and 7 (0.4 m^3 $year^{-1}$), and Scenarios 4 and 5 (26.5 m^3 $year^{-1}$). The total amount of carbon sequestered in above-ground biomass resulted as follows: (i) 1,931,921 Mg (scenario 1); (ii) 1,621,745 Mg ('freezing' scenarios); (iii) 1,937,786 Mg ('integral' scenarios); and (iv) 530,831 Mg ('more active' scenarios). Carbon sequestration increases in Scenario 1 (about 46,500 Mg $year^{-1}$), Scenarios 2 and 6 (about 36,000 Mg $year^{-1}$), and Scenarios 3 and 7 (about 47,000 Mg $year^{-1}$), while it decreases in Scenarios 4 and 5 (207 Mg $year^{-1}$). The total RS resulted as follows: (i) 0.040 (scenario 1); (ii) 0.042 ('freezing' and 'integral' scenarios); and (iii) 0.012 ('more active' scenarios). RS increase in Scenario 1 (31), in Scenarios 2 and 6 (30), and in Scenarios 3 and 7 (36), while they decrease in Scenarios 4 and 5 (70).

Factors impacting forest ecosystem services provision

Figure 20 shows the linear regressions between the FES provided and the IF considered, for each case study.

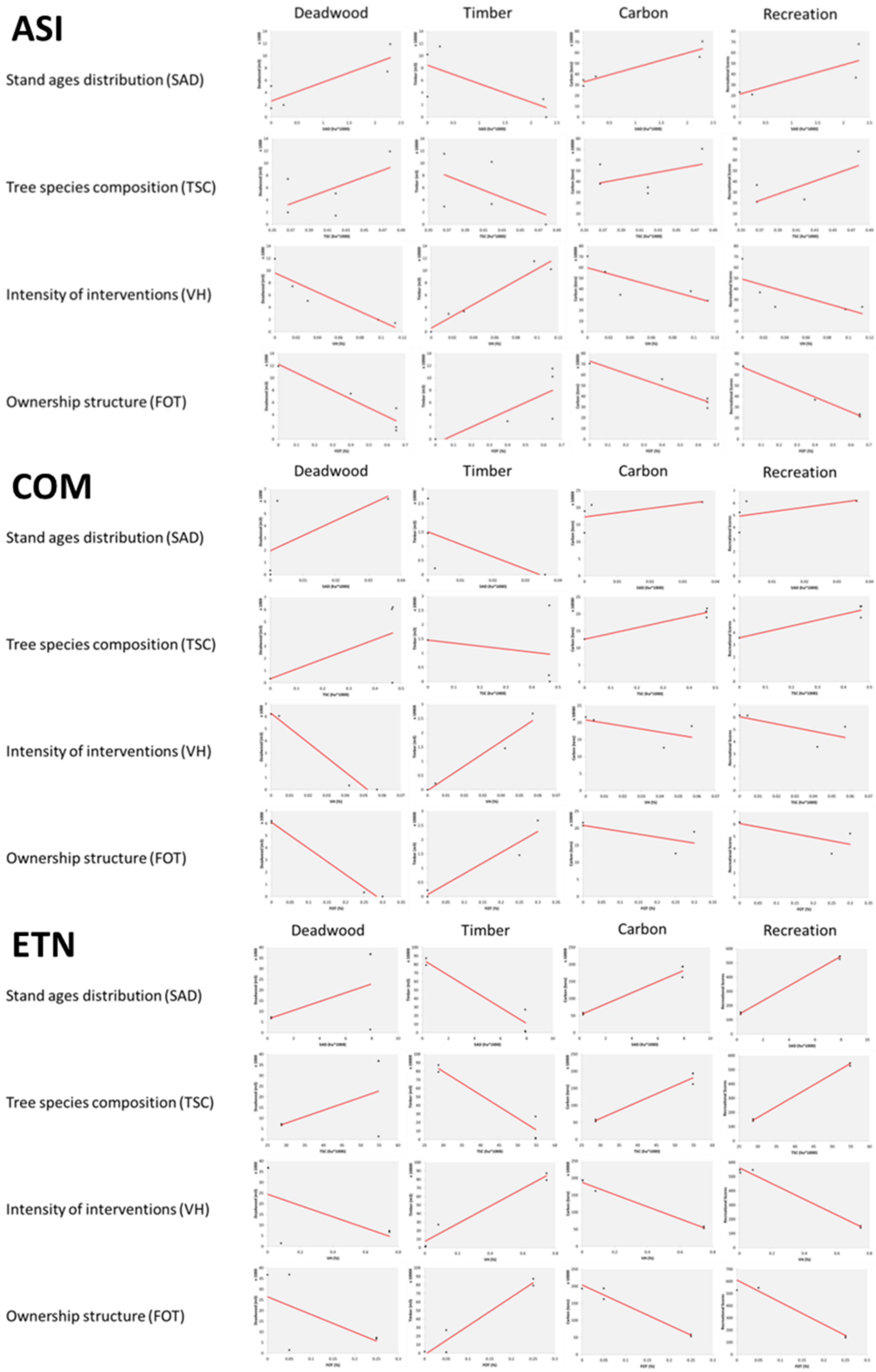


Figure 20: Linear regressions between forest ecosystem services and impact factors for the three case studies (from top to bottom, ASI, COM, and ETN) at the end of simulation period (30 years from now).

Results about the impact of different factors on FES provision in the ASI case study demonstrate that: (i) a stand age more than 150 years (SAD) positively contribute to the provision of timber, carbon and recreation; (ii) if the presence of broadleaves is more than conifers (TSC), then the amount of deadwood and the RS increase. On the other hand, the presence of broadleaves negatively impacts the provision of timber. It does not have a strong influence on carbon sequestration; (iii) if the ratio between the timber harvested and current growing stock is high (VH), then deadwood, carbon and recreation strongly decrease; the same occurs for the 'economic-oriented' ownership structure (FOT).

Results about the impact of different factors on FES provision in the COM case study reveal that: (i) ageing of stands (SAD) positively contributes to the provision of deadwood, carbon and recreation. By other hand, it negatively contributes to the provision of timber; (ii) if the presence of broadleaves is more than conifers (TSC), then the amount of deadwood and RS increase. On the other hand, the presence of broadleaves negatively impacts the provision of timber. It does not have a strong influence on carbon sequestration; (iii) if the ratio between the timber harvested and current growing stock is high (VH), then deadwood, carbon and recreation strongly decrease; the same occurs for the 'economic-oriented' ownership structure (FOT); and (iv) generally, the influence of IF is not so marked in relation to both carbon and recreation.

Results about the impact of different factors on FES provision in the ETN case study highlight that: (i) ageing of stands (SAD) positively contributes to the provision of deadwood, carbon and recreation. By other hand, it negatively contributes to the provision of timber; (ii) if the presence of broadleaves is more than conifers (TSC), then the amount of deadwood and the RS increase. On the other hand, the presence of broadleaves negatively impacts the provision of timber. It does not have a strong influence on carbon sequestration; and (iii) if the ratio between the timber harvested and current growing stock is high (VH), then deadwood, carbon and recreation strongly decrease; the same occurs for the 'economic-oriented' ownership structure (FOT).

3.4.4 Discussion and conclusions

This study offered an example of the integration between public participation (as in the scenario building phase) and the forest ecosystems simulation in three different landscapes in Italy. In this work, agent-based model (i.e. stakeholders preferences), decision-support systems for forest management (i.e. EFISCEN model), and socio-economic qualitative analysis have been integrated to simulate future-oriented scenarios of FES provision at landscape scale. Results concerning the FES provision among scenarios can be summarized as follows: (i) for ASI case study, the most balanced scenario is the 'High-quality trees-oriented silviculture' (scenario 4); for COM case study, 'Best case' and 'Prevailing *status-quo*' (scenarios 3 and 4, respectively) are competing scenarios for biodiversity conservation (reduced in scenario 3) and timber provision (reduced in scenario 4); and (iii) for ETN case study, the 'Integral' scenario (scenarios 3-7) is the most suitable scenario for improving the availability of all services, excepting than of timber provision. Generally, future-oriented simulations revealed that FES provision is mainly driven by: (i) the current conditions of forest stands, such as e.g. stand age distribution, turnover rate, regeneration capacity, and natural mortality rate; (ii) the silvicultural treatments applied, such as intensity and frequency of interventions, length of simulation period, and amount of timber released; and (iii) the timespan chosen for simulations (i.e. 30 years). Generally, the simulation period seems to be too short to evaluate the effects of those impact factors that are mainly related to forest growth and mortality (e.g. VH and SAD). Therefore, among all cases and scenarios, diversifying forest structure, reducing harvested volume, and prolonging rotation periods resulted as the most balanced choices to increase biodiversity conservation (in terms of amount of deadwood) and recreational opportunity (in terms of recreational scores). However, results from simulations did not offer a relative difference about the amount of carbon sequestered in above-ground biomass. Most probably, a timespan of 30 years is not particularly suitable to simulate a substantial mitigation of climate change effects (in terms of carbon sequestration). Considering the trade-offs between FES, timber provision is generally in conflict with the other services, especially in comparison with biodiversity conservation. This is partially explained by the fact that, in the case of timber provision, harvested volume is in generally removed, whilst deadwood (as proxy for biodiversity conservation) is the fraction of standing volume which is released. The implementation of different Owner Types in the simulations (behavioral models) revealed that FES provision (excepting that for timber) decreases if the presence and activity of owners increase (e.g. 'economic-oriented' FOT). Even in this case, the FOTs relative area sharing has no influence on carbon sequestration and recreational opportunities. Although EFISCEN model is a large-scale scenario model (using matrix cells larger than 1,000 ha), it proved to be suitable also for analyzing FES provision at landscape scale in e.g. Collemeluccio-Montedimezzo (approximately 700 ha). A further development of the model, specifically oriented to monitor land use changes is required.

At conclusion, the approach used in this work can be implemented to support IFM at landscape level, and extended to other contexts across the Mediterranen area. In particular, the engagement of local stakeholders in the scenario building phase was particularly suitable to understand local needs and behaviors while managing forest resources. In addition, simulating the availability of forest ecosystem services in the future was particularly interesting to understand the landscape potentialities in addressing the increasing search for benefits and in balancing people needs with intrinsic characteristics of ecosystems (i.e. resilience capacity). However, further developments are needed in order to operationalize IFM, mainly through considering land use changes and simulating other environmental disturbances.

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4

Assessing and monitoring the effects of forest management on ecosystem services provision

Forest management directly affects the development of forest stands over the time and across the space, thus influencing the provision of important goods and services. Moreover, intensity and frequency of forestry interventions affect the capacity of forest stands to face external impacts, such as e.g. climate change (drought, storms, insects outbreak), land use and cover transformations (neoformation forests), etc. As a consequence, assessing and monitoring the effects of management practices on resilience and stability of forest ecosystems is extremely important to ensure the continuous flow of benefits from these resources to local communities in the future.

In this chapter, the effects of forest management on ecosystem services are assessed and discussed through reporting the main outcomes from two case studies. The first one concerns the forest management in Protected Areas in Italy, which is mainly oriented to biodiversity conservation and tourism development. The second one deeper analyze the effects of alternative forest management systems on carbon sequestration (i.e. regulating service) in different mountain Forest Categories in Italy. In both cases, results mainly demonstrate that forest management can be oriented towards the 'resilience thinking' through integrating research with traditional forest knowledge, further involving local communities in decision-making processes, and implementing a more active forest management in degraded landscapes.

4.1 Overview on the role of forest management to improve biodiversity conservation and services provision

Management of ecosystem services (ES) is as complicated as managing ecosystems (Walters and Holling 1990). By an ecological point of view, ES management requires e.g. to (Kremen 2005): (i) identify the species and other entities that are ES providers, as well as characterizing their functional relationship; (ii) determine the various aspects of community structure that influence function in real landscapes; (iii) assess the key environmental factors influencing services the provision; and (iv) measure the spatio-temporal scale over which providers and services operate. To achieve sustainability and conservation goals, several forest management models have evolved over the time, and ranged from exploitative to close-to-nature-based approaches (e.g. Franklin *et al.* 2007). However, forest management itself requires strategies that have to be oriented to safeguarding essential ES (such as e.g. soil fertility and water quality), as well as fundamental supporting and cultural ES (such as e.g. carbon sequestration, biodiversity conservation, and recreational values) (Duncker *et al.* 2012). Accordingly, one of the most important questions for the future is how to manage the forest for timber production while conserving or improving other important ES (see e.g. Nelson *et al.* 2009). For example, restoring degraded forests can be one of the most effective strategies to improve ecological resilience, as well as to guarantee the provision of multiple services (Figure 21; Chazdon 2008).

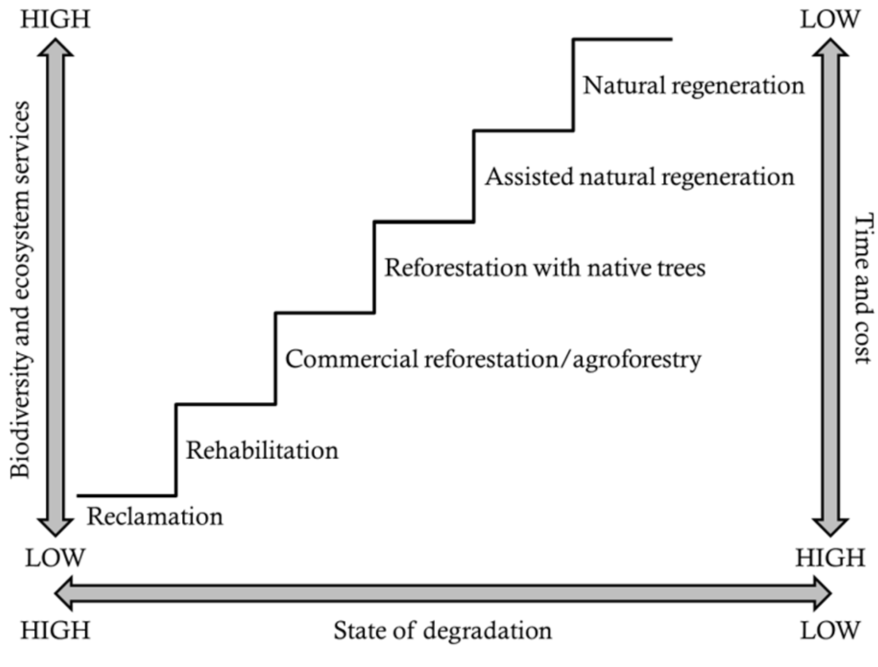

Figure 21: The restoration staircase. Depending on the state of degradation of an initially forested ecosystem, a range of management approaches can at least partially restore levels of biodiversity and ecosystem services given adequate time (years) and financial investment (capital, infrastructure, and labor) (Source: Chazdon 2008).

Several studies aimed at unrevealing the effects of forest management on biodiversity conservation (see e.g. Lindenmayer *et al.* 2006), climate change mitigation (Canadell and Raupach 2008; Sheller *et al.* 2011), and improvement of tourism opportunities (e.g. Boyd and

Butler 1996). Many other studies proved the negative effects of forest management on productivity (e.g. Helmisaari *et al.* 2011). Nevertheless, considerable efforts are needed to implement adaptive forest management and monitoring to (Lindenmayer *et al.* 2006): (i) better identify the impacts of logging operations and other kinds of management of biodiversity and ES; and (ii) quantify the efficacy of impact mitigation strategies and ways to improve practices where necessary. In the case of conflicting objectives, segregating forest management by forest service at the landscape level might be appropriate (e.g. Duncker *et al.* 2012). In the case of preserving biodiversity, this might include the retention of habitat elements such as coarse woody debris or veteran trees (Lindenmayer and Franklin 2002) and emulation of natural disturbances (see e.g. Farrell *et al.* 2000; Bengtsson *et al.* 2000). In addition, maintaining soil and water quality requires a reduction of nutrient losses, e.g., through harvesting of stemwood only instead of whole tree harvesting (Raulund-Rasmussen *et al.* 2008).

In order to preserve biodiversity and improve services provision, forest management is primarily called to balance economic revenues with ecological functionalities and local people needs (e.g. Kremen 2005). Accordingly, Butler and Koontz (2005) suggested the following requirements to make ES management operational: (i) collaborative stewardship; (ii) integrated scientific information sources; (iii) integrated social and economic information sources; (iv) adaptive management; (v) interagency cooperation; and (vi) sustainability. More broadly, these aspects move towards a strong integration between the ecosystem approach and sustainable management of forest resources, which can be realized through (Sayer and Maginnis 2005): (i) developing culturally and politically-appropriate mechanisms for defining both the production goals of immediate concern to local resource user groups and the environment and development goals of the wider society; (ii) developing programmes to negotiate the institutional arrangements for integrated landscape management and define processes that will maximize positive synergies and minimize negative synergies between forest and other land uses; (iii) clarifying fair and workable institutional, policy and legal arrangements with respect to the rights and responsibilities of forest ownership and use; (iv) aligning and reforming resource access prices, payments for environmental services and fiscal constraints and incentives to encourage resource sustainability and the internalization of all externalities associated with particular resource use patterns; and (v) developing participatory monitoring, evaluation and review mechanisms that will allow iterative improvement in land use allocation and management processes in response to new scientific information, changing environmental, social and economic conditions and the experience gained from landscape management.

4.2 Case study 4: Managing forests in Protected Areas to maximize biodiversity conservation and services provision[7]

4.2.1 The context

In the frame of ES, biodiversity is intimately linked to the ecosystems' functionality and human wellbeing, in the following ways: (i) biodiversity has a multilayered relationship with the other ES – as a regulator of ecosystem processes, as a service in itself and as a good (Mace *et al.* 2012); (ii) the loss of biodiversity is one of the most influencing drivers of ecosystem change, in terms of primary production and decomposition (Hooper *et al.* 2012); and (iii) the loss of biodiversity-dependent ES is likely to accentuate inequality and marginalization of the most vulnerable sectors of society (Díaz *et al.* 2006). These key-points have been largely reviewed (e.g. Cardinale *et al.* 2012), and used as basis to support research proposals targeting at biodiversity conservation worldwide (see e.g. "The IUCN Red List of Ecosystems"; Keith *et al.* 2013). Moreover, several regulatory frameworks concerning the safeguard of biodiversity and ecosystems functionality are available at global (see the CBD "2020 Aichi Target"; www.cbd.int/sp/targets), European (see the "EU Biodiversity Strategy"; European Commission 2011), and Italian level (see *Strategia Nazionale per la Biodiversità* [Italian National Biodiversity Strategy]; Andreella *et al.* 2010).

In this context, conservation strategies have to develop from management objectives around defined needs and explicit values (Perrings *et al.* 2010), which might be anywhere on a spectrum from strictly utilitarian (e.g. to maximize carbon sequestration) to completely cultural values (e.g. to conserve a rare endemic species) (Mace *et al.* 2010). As a consequence, taking into account that the management of ecosystems (forests included) is consistent with the scale at which it is implemented (Schneiders *et al.* 2012), the Protected Areas' Network (PAN) has a key role for handling the global biodiversity conservation (see e.g. Reid 1998; Bruner *et al.* 2001, etc.), as well as the provision of other services (Hockings 2003). As examples, PAN covers more than 12% of the Earth's land area, of which more than 7.5 million ha belong to forest biomes (Chape *et al.* 2005), and in Italy, forests and OWL that are included in PAN at Country level cover more than 1.5 million ha (approximately 50% of the total PAN).

More specifically, while securing ES is vital to human wellbeing, current intensive management may also potentially undermine the capacity of forests to sustain this production in the future (Bennett and Balvanera 2007; Fisher *et al.* 2009), as well as to meet emerging demands for new goods and services (Canadell and Raupach 2008). The combination of anthropogenic impacts and demands on forests coupled with global change, suggests that compounded perturbations and ecological threats will become more common (Paine *et al.* 1998). As a consequence, forest management faces a substantial challenge if the capacity of forests to provide valued ecological goods and services in the future is to be maintained (Rist and Moen 2013). Considering these emerging challenges, the conventional management methods in forestry need to be revised by implementing innovative and integrated approaches

[7] Vizzarri *et al.* (2014, *submitted*).

according to both multi-functionality and sustainability of the forest resources management mandate (for the Italian context, see Ciancio and Nocentini 2011; Corona and Scotti 2011; Marchetti 2011). In contrast to the widely held view that forest management should provide services for human uses, maintaining forest functionality in the context of a changing environment may require focusing on the forests themselves and on strategies to reduce their vulnerability to increasing stress. Forest conditions that best meet the demands for ES must first be defined, and then, the management pathways that allow the forest to be adapted to this target state need to be identified. However, targeting forest sustainability is not an easy task, as it depends on legacies from past management as well as uncertainties in future climate. As examples, the adaptive management (Holling 1978; Walters 1986), the ecosystem approach (CBD 2004), and the "resilience thinking" (Gunderson 2000; Folke 2006; Walker and Salt 2006) can be considered as the most suitable forestry approaches dealing with sustainability challenges (Rist and Moen 2013).

4.2.2 Objectives and methodology

A 20-pages questionnaire on the role of FES in socio-economic and planning contexts was prepared and then submitted to a target-group of National and Regional Parks (NRPs) in Italy. 15 NRPs were selected by a group of experts in the forest management and planning fields at national level, according to the following criteria: (i) regional representativeness (more than 80% of the Regions should be represented); and (ii) forest area inclusiveness (more than 25% of the total forest area should be included). Questionnaires contained different "closed-questions" and were divided into seven sections, as follows: (i) general information; (ii) main FES provided; (iii) local stakeholders-FES relationship; (iv) governance instruments currently available; (v) main factors influencing decision-making processes; (vi) linkages between decision-making processes and research activities; and (vii) FES relevance in forest management. Along the questionnaire, the interviewees were called to answer by using ranking scales or true/false options. Then, answers were separately analyzed by questionnaire's section. For further details about the questionnaires' structure and the methodology adopted for the analysis of the results, the reader is referred to Appendix 2.

4.2.3 Results

Questionnaires were filled in by 15 NRPs in Italy, and the related characteristics (see Table 25) demonstrate that the selection criteria were fulfilled, in terms of Regions represented and forest area covered.

Table 25: List of surveyed National and Regional Parks.

Park name	IUCN classification	Covered Regions	Covered Provinces	Covered Municipalities	Total area	Forest area (%)	Managed forest area	Main Forest Category
"Pineta di Appiano Gentile e Tradate" Park [*Parco Pineta di Appiano Gentile e Tradate*]	Ib	1	2	15	4,828	0.72	1.000	Scots pine and black pine forests
"Mont-Avic" Natural Park [*Parco Naturale Mont Avic*]	Ib	1	1	2	5,800	0.28	NA	Scots pine and black pine forests
"Vesuvio" National Park [*Parco Nazionale del Vesuvio*]	II	1	1	13	8,482	0.44	0.541	Coniferous plantations
"Prealpi Giulie" Natural Park [*Parco Naturale delle Prealpi Giulie*]	Ib	1	1	6	9,402	0.49	0.782	European beech forests
Aurunci mountains Natural Park [*Parco Naturale dei Monti Aurunci*]	Ib	1	2	10	19,374	0.59	0.947	Hop-hornbeam and flowering ash mixed forests
"Dolomiti Bellunesi" National Park [*Parco Nazionale delle Dolomiti Bellunesi*]	II	1	1	15	32,000	0.69	1.000	European beech forests
"Foreste Casentinesi, Monte Falterona, e Campigna" National Park [*Parco Nazionale delle Foreste Casentinesi, Monte Falterona e Campigna*]	II	2	3	10	36,800	0.87	0.781	European beech forests
"Abruzzo, Lazio e Molise" National Park [*Parco Nazionale d'Abruzzo, Lazio e Molise*]	II	3	3	24	50,000	0.58	0.655	European beech forests
Adamello Natural Park [*Parco Naturale dell'Adamello*]	Ib	1	1	18	51,000	0.46	0.062	Other broadleaved forests
Adamello-Brenta	Ib	1	1	40	62,050	0.81	1.000	Norway

Park name	IUCN classification	Covered Regions	Covered Provinces	Covered Municipalities	Total area	Forest area (%)	Managed forest area	Main Forest Category
[*Parco Naturale Adamello-Brenta*]								spruce forests
“Monti Sibillini” National Park [*Parco Nazionale dei Monti Sibillini*]	II	2	4	19	69,439	0.42	0.956	European beech forests
“Majella” National Park [*Parco Nazionale della Majella*]	II	1	3	39	74,100	0.66	NA	European beech forests
“Gargano” National Park [*Parco Nazionale del Gargano*]	II	1	1	18	121,400	0.52	1.000	European beech forests
“Stelvio” National Park [*Parco Nazionale dello Stelvio*]	II	4	4	23	130,700	0.29	0.762	Norway spruce forests
“Pollino” National Park” [*Parco Nazionale del Pollino*]	II	2	3	56	192,000	0.58	0.140	European beech forests

The following insights can be denoted from the results from section 1: (i) the total surveyed area is 31.4% of the total NRPs area (867,375 ha vs. 2,760,337 ha); (ii) the total surveyed forest area is 471,181 ha, which is 4.5% of the total forest area in Italy; and (iii) the average forest area is approximately 56% of the total PAN, of which 69% is managed (totally managed for 31% of NRPs). By the administration point of view, the NRPs selection included more than 72% of the total number of Regions and autonomous Provinces in Italy.

Figure 22 shows the relevance of FES in NRPs, as resulted from section 2 of the questionnaires.

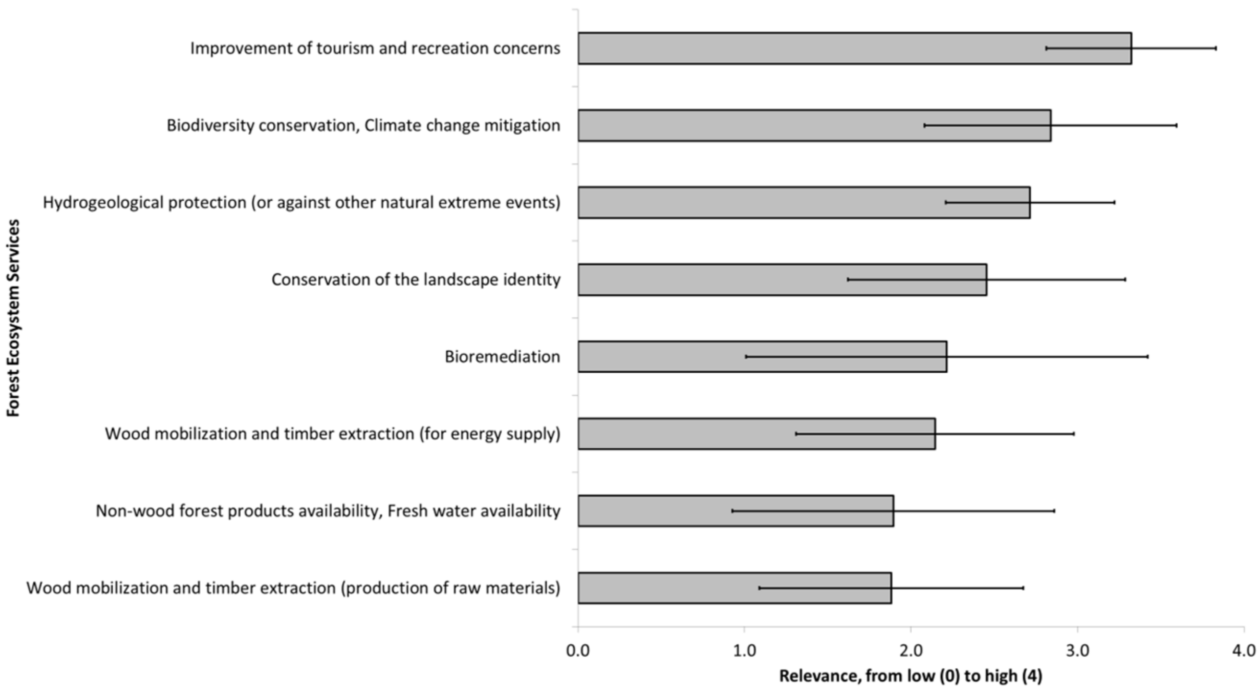

Figure 22: Bar chart reporting the FES relevance (grouped by Division) in surveyed NRPs. Values vary from very low (0) to very high (4) relevance. Errors bars refer to SD values.

The survey on the FES relevance resulted as follows: (i) the improvement of tourism and recreation concerns is considered as fundamental service (3.3±0.51), thus showing higher relevance in comparison with e.g. the provision of raw materials (wood and fibers, among the others) (1.9±0.79); (ii) the enhancement of interactions with forest ecosystems (here defined as conservation of the landscape identity) is considered less important (2.5±0.83); (iii) the maintenance of ecosystem processes and functions (including biodiversity conservation and habitat protection) is considered as a very important forest service (2.8±0.76); (iv) the hydrogeological protection and bioremediation are considered averagely important (2.7±0.51 and 2.2±1.21, respectively); and (v) mainly the biomass-based energy sources are considered averagely important (2.1±0.83).

Very high variability between results refers to the relevance values of both the bioremediation service (SD=1.20), and non-wood forest products (NWFPs) and fresh water availabilities (SD=0.93). On the contrary, very low variability refers to the relevance values of both hydrogeological protection and the improvement of tourism and recreation concerns (SD=0.51 for both). These aspects demonstrate that: (i) most probably there was a partial lack of knowledge (or misunderstanding) of some of the ES definitions, particularly from forests, while assessing their relevance; and (ii) some of the ES were not always considered as services primarily from forests, thus not assessed in the same way by all of the respondents.

Figure 23 reports the impacts of different stakeholder typologies on the whole set of FES (section 3 of the questionnaire).

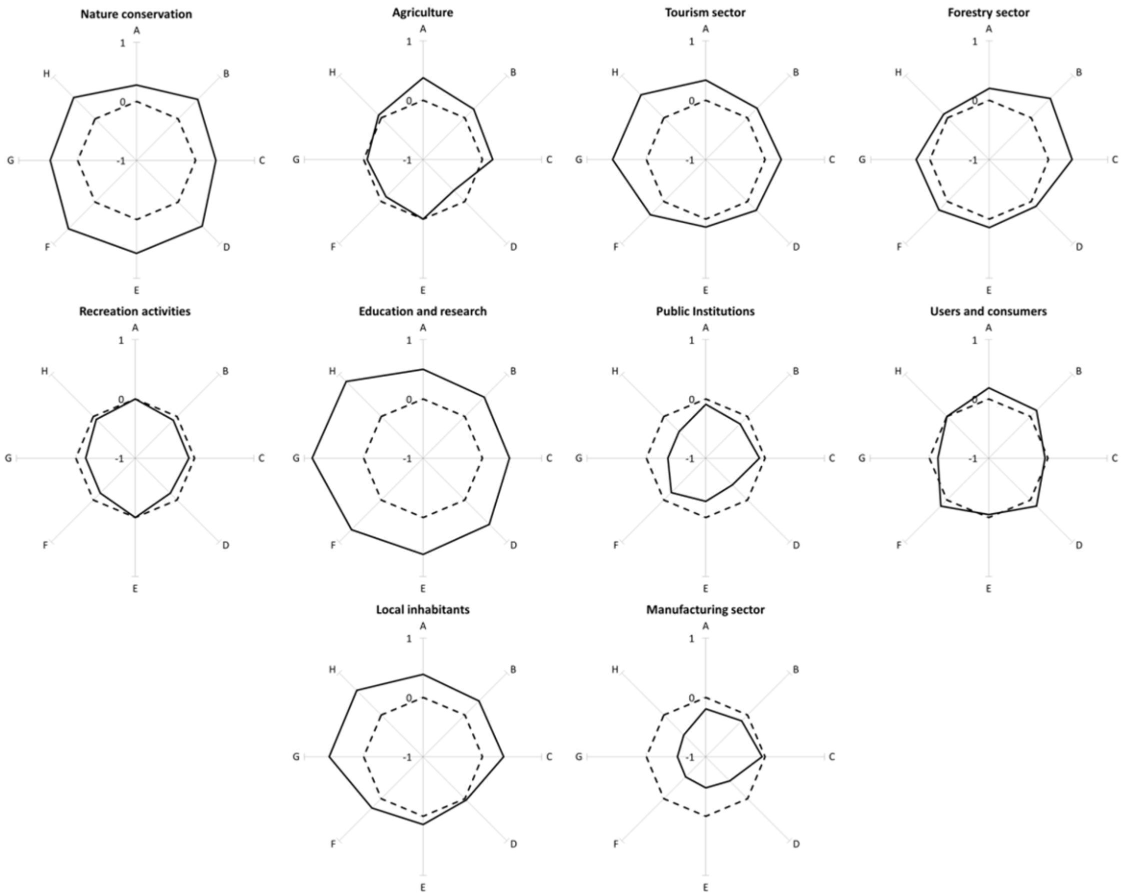

Figure 23: Spider charts showing the different impacts of stakeholder typologies on FES, lettered from A to H as follows: NWFPs/fresh water availability (A); Production of raw materials (B); Timber extraction for energy supply (C); Bioremediation (D); Hydrogeological protection (E); Biodiversity conservation and climate change mitigation (F); Improvement of tourism and recreation concerns (G); and Conservation of the landscape identity (H) (see Appendix 2 for the nomenclature). Values range from -1 (negative impact) to 1 (positive impact). Dashed lines represent the 0 value.

Results from section 3 of the questionnaires reveal that: (i) NGOs, European Union, Park Management Authority, and other public bodies, acting as nature conservationists, have a positive impact on the whole set of FES, ranging from 0.28 for the availabilities of NWFPs and fresh water to 0.64 for the conservation of landscape identity; (ii) agriculture and pasture have few positive impacts (e.g. on NWFPs and fresh water availabilities, 0.38), and more negative impacts, especially on bioremediation (-0.26) and on biodiversity conservation and climate change mitigation (-0.12); (iii) the tourism sector has a positive role for all FES, especially with regards to the improvement of tourism and recreation concerns (approximately 0.56); (iv) the forestry sector generally has no impact on FES at all, thus registering the highest positive values in the cases of the production of raw materials (0.48) and biomass-based energy supply (0.40); (v) recreation activities (e.g. hunting , mushrooms picking, skiing, etc.) are generally seen as relatively negative influencing factors on FES provision (between -0.07 and -0.17 for the largest part of FES); (vi) education and research activities are considered as the most influencing driver for improving the FES provision, with values from 0.46 (production of raw materials and biomass-based energy supply) to 0.87 (improvement of tourism and recreation concerns); (vii) Public Institutions (intended here as Army-related activities) are considered as limiting factors for the FES provision (e.g. -0.36 for bioremediation); (viii) local users and farmers averagely have no impact on FES provision (values are around 0 for all FES); (ix) local inhabitants are seen as drivers for the FES provision, especially with regards to the improvement of tourism and recreation concerns and the conservation of landscape identity (0.58 for both); and (x) the manufacturing sector is the most negative influencing factor (i.e. the most limiting agent) for the FES provision, with values ranging from -0.05 for biomass-based energy supply to -0.52 for biodiversity conservation and climate change mitigation.

Figure 24 shows the presence/absence of the main governance instruments at work in the surveyed NRPs area.

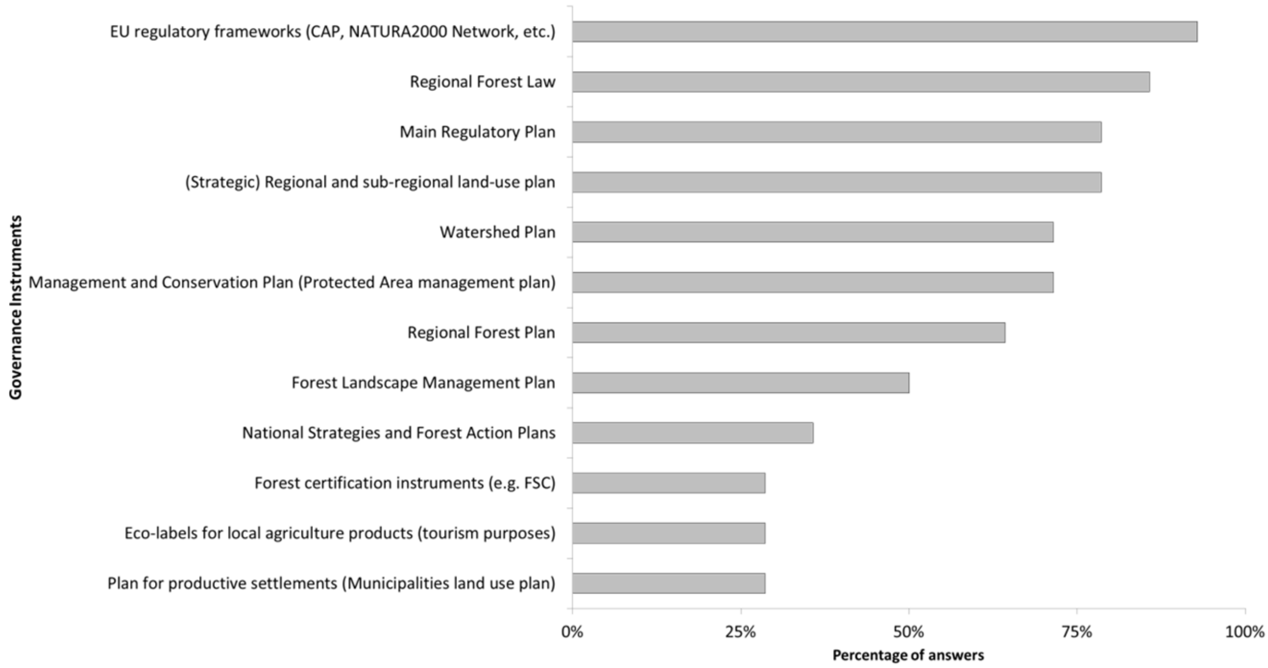

Figure 24: Percentage of NRPs where governance instruments are currently at work.

Results from section 4 of the questionnaires show that some of the most important governance instruments with regards to biodiversity conservation (*gen.* nature conservation) and FES provision are at work in more than 50% of the cases. Downscaled by implementation level, they are e.g. "EU regulatory frameworks" (93% of respondents), "Regional Forest Law" (86% of respondents) and "Regional Forest Plan" (64% of respondents), "Watershed Plan" (71% of respondents) and "Forest Landscape Management Plan" (50% of respondents), and "Management and Conservation Plan" (71% of respondents). By other hand, other important governance instruments, such as "National Strategies and Forest Action Plans", "Forest Certification Instruments", and "Eco-labels for local agricultural products" are widely not yet available or implemented (36%, 29% and 29% of respondents, respectively).

Figure 25 shows the results about the role of different factors (e.g. local stakeholders, FES analysis, etc.) in decision-making processes and forest management concerns (section 5 of the questionnaire).

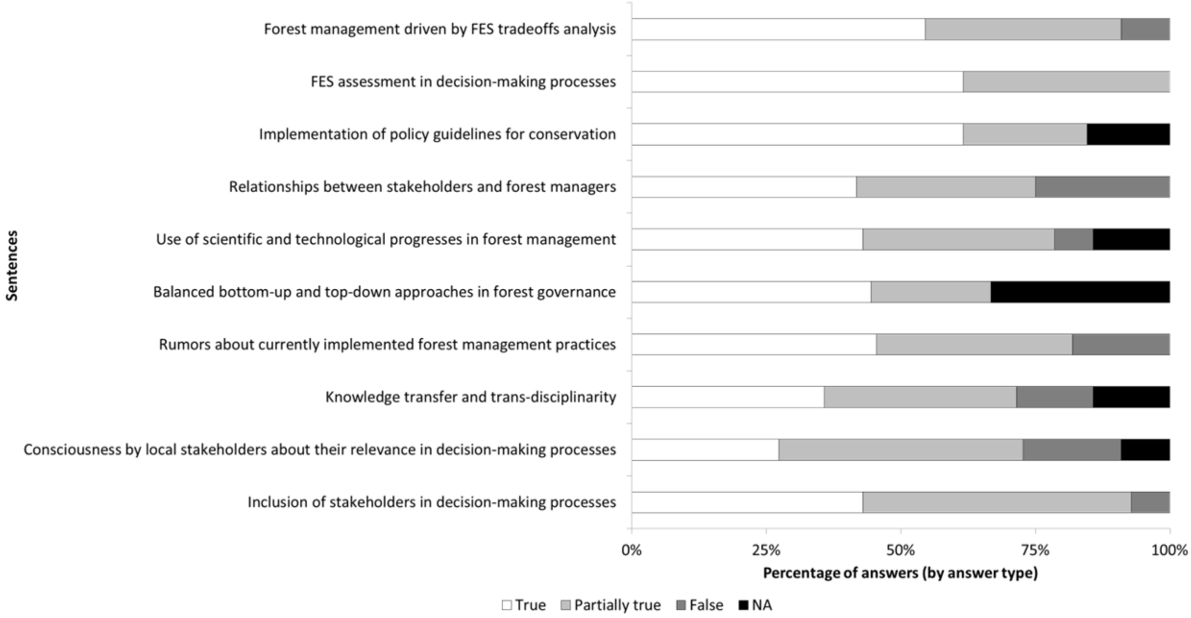

Figure 25: Bar chart showing the relative percentage of respondents to several sentences regarding the role of different factors (local stakeholders, FES analysis, etc.) in decision-making processes.

Results from section 5 of the questionnaires can be summarized as follows: (i) FES assessment is included in decision making processes, and related tradeoffs analysis is considered in implementing forest management (true for 62 and 55% of respondents, respectively); (ii) nature conservation is implemented by adopting specific policy guide-lines (true for 62% of respondents); (iii) stakeholders are partially engaged in decision-making processes (true for 43% of respondents), for which they are recognized as not relevant (see "Consciousness by local stakeholders about their relevance in decision-making processes", which is true for 27% of respondents), as well as they have rather few relationships with forest managers (true for 42% of respondents); (iv) bottom-up and top-down approaches in forest governance are not completely balanced (true for 44% respondents); (v) knowledge-transfer and trans-disciplinarity are generally missing (true for 36% of respondents); and (vi) the scientific and technological progresses or advances are not always taken into account in forest management (true for 43% of respondents). These results have a high level of uncertainty. In fact, the "False/Not Available (NA)" answers correspond to 13% of respondents, averagely. Moreover, the "Partially true" answers (36% of respondents, averagely) do not offer more details to deeper analyze the results (and the differences with regards to the "True" answers).

Figure 26 shows how the research is linked to the management of forests and their services within the surveyed NRPs area.

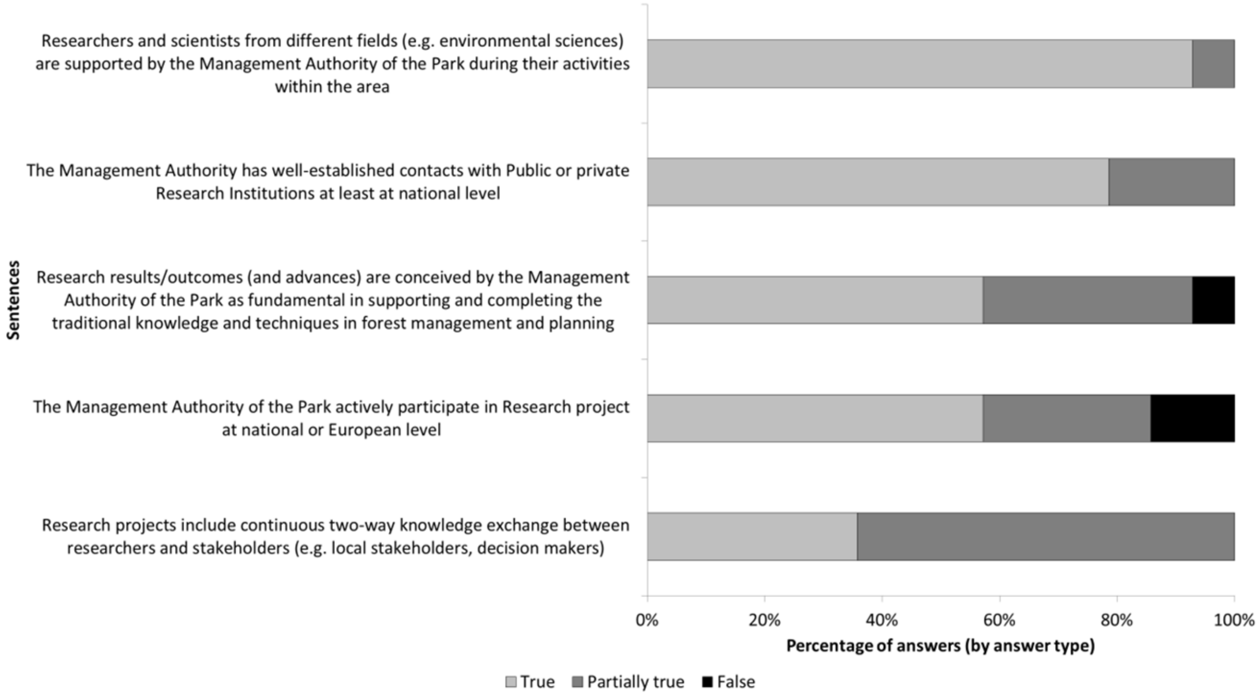

Figure 26: Bar chart showing the relative percentage of respondents to several sentences regarding the role of research (related activities, results and advances) in decision-making processes.

Results from section 6 of the questionnaires can be summarized as follows: (i) the Management Authorities support researchers and their activities within the Park area (true for 92% of respondents), as well as they are in cooperation with several Research Bodies at national level (true for 78% of respondents) and participate to different research projects (true for 57% of respondents); (ii) although the Management Authorities conceive as fundamental the support of research outcomes for improving practical forest management (true for 57% of respondents), research projects do not consider the two-way exchange between stakeholders and decision-makers (true for 35% of respondents).

Figure 27 reports the ranking values of three important FES-Forest Management linkages (section 7 of the questionnaire), such as (i) the priority for management guide-lines, (ii) difficulty in quantifying and valuing the service, and (iii) the relevance for local communities (in terms of expected benefits).

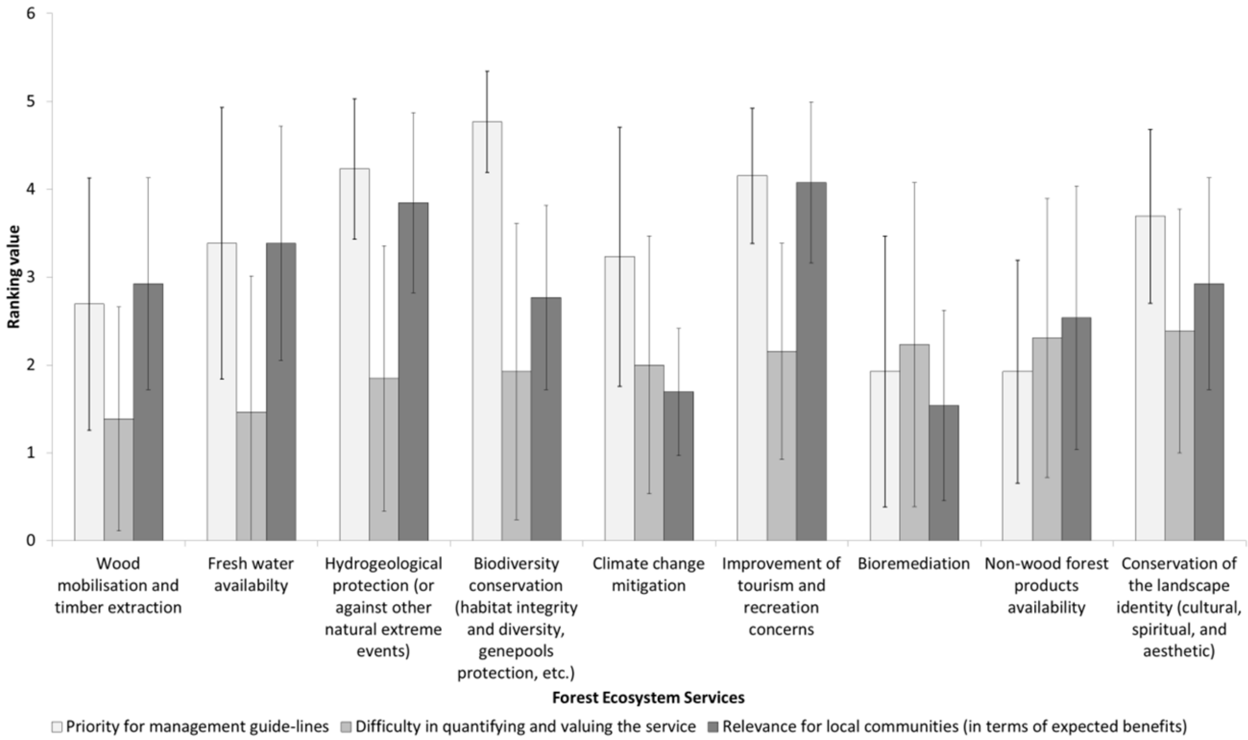

Figure 27: Bar chart reporting the ranking values in assessing FES priority, difficulty and relevance in forest management. Error bars represent the Standard Deviation (SD) values.

According to the results about the priority for management guidelines, three FES have the highest values (more than 4, high priority), such as the biodiversity conservation (4.76±0.57), the hydrogeological protection (4.23±0.79), and the improvement of tourism and recreation concerns (4.15±0.76). On the other hand, the NWFPs availability and bioremediation services show the lowest priority values (1.92±1.26 and 1.92±1.54, respectively).

Analyzing the results about the relevance for local communities, the improvement of tourism and recreation concerns has the highest value (4.07±0.91). Adversely, the bioremediation shows the lowest relevance value (1.58±1.08). Considering the difficulty in quantifying and valuing the service, respondents used very low ranking values. These results indicate that there is no very high difficulty to have a quantitative or qualitative assessment (measure) of a given FES, including its economic evaluation. For example, the conservation of landscape identity was ranked as the most difficult FES being assessed. Of course, this element (i.e. "relevance") shows a higher variability (SD between 1.23 and 1.84) in comparison with "priority" (SD between 0.57 and 1.54) and "difficulty" (SD between 0.72 and 1.49) ones.

4.2.4 Discussion and conclusions

Our survey on the implementation of ES into forest management in NRPs in Italy offered a wide view about the conservation policies in act and the management of forest ecosystems in general, as well as on the involvement of stakeholders and local communities in decision-making discourses. The most important key issues on managing forests for maximizing services provision are hereinafter discussed.

Relevance for biodiversity conservation

The conservation of biodiversity, the improvement of tourism and recreation concerns, and the hydrological protection are the most relevant FES both for forest management purposes and local community needs (see Figure 22). They also have the highest priorities towards effectively implementing management guidelines (see Figure 27). These results may depend on the following factors: (i) biodiversity conservation is the primary objective of the Protected Areas establishment, because their original mandate and key role both in Italy (Duprè *et al.* 2013) and globally (Chape and Mulongoy 2004); (ii) the tourism sector and recreation activities are two of the key economic drivers for improving the wellbeing of people living within the Protected Areas boundaries (see e.g. Naughton-Treves *et al.* 2005); and (iii) the regulation of hydrological regimes to protect human infrastructures against floods, runoff erosions, avalanches or other natural hazards is one of the most important forest functions, especially if regulated by specific laws and restrictions as in Italy (Motta and Haudemand 2000; Scarascia-Mugnozza *et al.* 2000). Adversely, the provision of timber, fibers, and other forest products is considered as less relevant (see Figure 22). This is partially explained by the fact that the largest part of regulatory frameworks of NRPs in Italy strictly limits the use of forest resources for economic purposes (including timber extraction and transformation), at least in their core areas for biodiversity conservation (see Schneiders *et al.* 2012). In this sense, our results confirm that although provisioning services generate economic benefits for local population living in Mediterranean area (for a complete review, see Croitoru 2007), they are generally in conflict with other services, especially with biodiversity conservation (see e.g. Raudsepp-Hearne *et al.* 2010).

Linkage between services and forest management

Results from section 5 of the questionnaires reveal the following insights: (i) although FES are assessed for decision-making objectives and their trade-offs considered while managing forests (see Figure 25), there are no specific information about the level of detail in quantifying FES, neither on the methodologies applied and the quality of data used for such a purpose (see Appendix 2, Section 5); (ii) stakeholders are not really involved in decision-making processes, so that they do not consider their opinions as relevant for forest management; and (iii) there is not always a well-balanced top-down and bottom-up approach (see Figure 25).

Knowledge transfer to local communities

Although the Management Authorities of NRPs use research activities as a support for monitoring biodiversity and managing forests for ES maximization, there is no knowledge transfer to local communities (see Figure 26). It means that research outcomes and advances are not delivered to and shared with people, such as stakeholders or inhabitants, despite 'Education and Research' are considered the most positive drivers for the FES provision (see Figure 23).

Relationship between forest ecosystem services and local stakeholders

Understanding the role of stakeholders in the contexts of forest management and the FES provision is extremely important because the value of ES upon their views and needs (Vermeulen and Koziell 2002), thus improving the coupled human-environmental systems relationship (see e.g. Hein *et al.* 2006). Moreover, adaptive forest management properly builds on the sharing of management responsibility between different sets of stakeholders operating at different levels (Folke *et al.* 2005). From the results (Figure 23), it is clear that the relationship between local stakeholders and the provision of FES depends on at least two main factors, as follows: (i) the identified stakeholder typology (public bodies, inhabitants, and private companies); and (ii) the stakeholder's behavior with regards to the FES provision (provider or consumer, source or beneficiary). About the stakeholder typology, 'nature conservation', 'education and research', 'tourism' and 'forestry' sectors show the highest positive influences (seen as public bodies and institutions), especially with regards to biodiversity conservation, habitat integrity, and the maintenance of cultural and spiritual values FES. Adversely, inhabitants and private companies are seen as impacting on the FES provision. About the stakeholders' behavior, the providers ("nature conservation" and "education and research") obviously are considered as drivers for the FES availability. On the other hand, the users ("local inhabitants", "recreation activities", and "manufacturing sector") are considered as barriers for the FES availability.

Challenges for managing forest services and conserving biodiversity

The survey on NRPs in Italy allowed investigating whether the currently implemented forest management is oriented towards the biodiversity conservation and the maximization of FES provision. Considering that the human-induced effects on forest ecosystems and biodiversity have been largely proved (see e.g. Vitousek *et al.* 1997), current forest management practices and silvicultural interventions should be translated from a monetary-centered into more sustainable and holistic approaches (Ciancio and Nocentini 2011; Marchetti 2011). Especially in fragile and degraded forest environments (i.e. in mountain areas) in changing times (Lindner *et al.* 2010), forest management is called to ensure forest health, vitality and stability over the long run, in order to maximize the ecosystem functioning and the provision of the whole set of goods and services (see e.g. Folke *et al.* 2004). In Italy, the network of NRPs can play an active role in conserving forest biodiversity and preserving the delivery of all FES (see e.g. Schirpke *et al.* 2014; Marchetti *et al.* 2012(a)). Having a large portion of the Country under nature conservation regimes is particularly suitable if considering the human-induced effects on land use change at the expenses of natural environments (see e.g. Corona *et al.* 2012;

Marchetti *et al.* 2012(b)). By other hand, what can be the challenge outside Protected Areas? Searching for the best FES trade-offs requires the adoption of the "resilience thinking" in forest management, which evolved from the concept of "sustainability" and "ecosystem-based" approaches (for a review of the three approaches, see Rist and Moen 2013). Managing forests according to their resilience practically aims at maintaining the system function, structure, and feedbacks (identity), through including the following steps: (i) characterize the essential elements influencing forest health and productivity (e.g. so called slow variables), (ii) provide a means of translating these into a set of conditions or processes by which management goals may be set and achieved, and (iii) implement methods for assessing progress towards these goals, namely metrics such as management indicators and reference points (Rist and Moen 2013).

As also outlined by the results, forest management has to be built on perceptions and needs of local communities and stakeholders through a bottom-up approach in decision-making processes (not currently at work, at least in NRPs). Specifically, the top-down, technocratic welfare economic approach provides few opportunities for stakeholders to contribute to the decision process of complex and socially contentious problems beyond expressing individual preferences via monetary bids in response to the various valuation methods (Chee 2004).

In agreement with both the international and national commitments (Andreella *et al.* 2010), the development of a National Ecosystem Assessment Framework in Italy – as already adopted in other Countries around the world (see e.g. Daily *et al.* 2013; UK-NEA 2011) – is urgent and needful for better orienting decisions in a sustainable way, especially within the forestry sector. This tool can provide a unique framework, which can be used for mapping and assessing (in both ecological and economic ways) FES from the local to a national level, thus contributing to the monitoring of biodiversity conservation and resilience of forest ecosystems.

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4.3 Case study 5: The effects of forest management on carbon sequestration: the case of mountain Forest Categories in Italy[8]

4.3.1 The context

Forest ecosystems have a fundamental role in regulating climate (Bonan *et al.* 2008), and in mitigating related changes (Canadell and Raupach 2008), from global to local scale. Forest ecosystems absorb approximately 1.68 Mg C ha^{-1} $year^{-1}$ at European scale (Pan *et al.* 2011), whereas forest C stock in Italy ranges between 16.3 and 88.7 Mg C ha^{-1} (Gasparini and Tabacchi 2011). However, although European forests are found to have a positive Net Ecosystem Productivity (NEP), this sink tends to decrease in Countries whit older and still ageing forests (Bellasen *et al.* 2011). Nabuurs *et al.* (2013) pointed out that "carbon sink saturation seems to be quite imminent in managed European forests" and that "these forests are reaching a dynamic equilibrium with the current intensity of management, tree species and age-class distribution". Despite in EU forest biomass will almost certainly expanding in the forthcoming decades (Rautiainen *et al.* 2010), in the long run forest management can significantly contribute to EU's effort to control its net emissions of greenhouse gasses (Peters *et al.* 2009). Particularly in temperate forests, forest carbon stocks typically increase with age until becoming relatively stable after about 100–150 years, while net ecosystem carbon balance (NECB) often peaks much earlier and gradually declines to near zero (Pregitzer and Euskirchen 2004; Bradford and Kastendick 2010; Williams *et al.* 2012). Similarly, Luyssaert *et al.* (2010) showed that forests between 15 and 800 years of age can accumulate carbon and have a positive NEP (including trees and soil), but there is an age-related decline, which after a number of years, depending on the type of vegetation and environment, reaches an equilibrium. In summary, the forest potential in sequestering carbon (in terms of increasing, or at least maintaining, the carbon stock into the future) is mainly regulated by climate change conditions, disturbances (Seidl *et al.* 2014; Lindroth *et al.* 2009: van der Werf *et al.* 2010), age structure, and tree species composition.

Forest management has variable effects on carbon stocks over time (see e.g. Matthews *et al.* 2014), as it contributes to increase the productivity of forest stands, and as a consequence the carbon accumulation (see e.g. Kaipainen *et al.* 2004: Harmon and Marks 2002). Carbon sequestration process is continuing at significant levels under current conditions and, in principle, could be 'managed'. Several studies aimed at separating the impact of changes in climate and forest management on the future carbon balance of European forests (see e.g. Nabuurs *et al.* 2002; Eggers *et al.* 2008; Pussinen *et al.* 2009), but relatively few studies have examined the differences of biomass carbon and net carbon sequestration ability related to forest stand age among different forest types (Chen *et al.* 2013). A number of recent studies on forest carbon sequestration have focused on in situ C storage, which has been investigated on varying scales, for different ecosystems and regions (e.g. Liski *et al.* 2000, Liski *et al.* 2003; Thornley and Cannell 2000; Lee *et al.* 2002; Pussinen *et al.* 2002; Dean *et al.*, 2004; Howard *et al.* 2004; Pregitzer and Euskirchen 2004; Backeus *et al.* 2005 and Lasch *et al.* 2005). Results *inter*

[8] Source: Vizzarri *et al.* (2015, *submitted*).

alia point towards considerable effects of management on forest carbon storage, however, quantitative impacts and economic feasibility vary. In order to improve carbon sequestration potentials, several authors suggested to handle the forest management according to the diversity of forest types and to the availability of other ecosystem services (Nabuurs *et al.* 2013), as well as to increase the harvesting rates or reduce the rotation period (Pussinen *et al.* 2009; Seidl *et al.* 2014). Enhancing the positive effects of forest management on carbon sequestration would require a systematic and coordinated effectiveness of management across forest areas, by combining increased harvesting in some areas, and conservation and enrichment of carbon stocks in other areas (including afforestation options) (e.g. Nabuurs *et al.* 2008; Matthews *et al.* 2014). Currently, there has been limited exploitation of the potential for such options.

According to Lindner *et al.* (2008), climate change has negative impacts on carbon sequestration of Mediterranean forests in the following ways: (i) for forests located in Mediterranean bioclimate, drought on forest growth and productivity will also affect carbon sequestration rates and the net carbon balance will be strongly affected by disturbances, especially by projected increases in frequency and intensity of forest fires; (ii) for forests located in mountain environments and for the second half of 21st Century, the increasing respiration rates and frequent disturbances at low elevation sites are projected and therefore the sink function will decrease and forests may become a carbon source (Karjalainen *et al.* 2002; Thürig *et al.* 2005; Zierl and Bugmann, 2007; Seidl *et al.* 2008a; Seidl *et al.* 2008b); (iii) for forests located in the temperate-continental zone, in the short term it was argued that they may be a source rather than a sink for atmospheric carbon as the relative distribution of carbon among ecosystem components adjusts in response to changing climate conditions (Vucetich *et al.* 2000). Maintaining, restoring and establishing ecologically and economically viable forest communities will require multiple strategies, including efforts to maximize carbon storage in standing forests and wood products (Resco de Dios *et al.* 2007). Management strategies to mitigate climate change can be divided into three approaches (Brown1997; Eggers 2002), such as: (i) conservation management (preventing emissions and conserving forest carbon pools); (ii) storage management (increasing carbon stocks); and (iii) substitution management (maximizing the time carbon is sequestered as wood). It is also well argued that in Mediterranean area a proper management might increase carbon sequestration under climate change (Seidl *et al.* 2008a; Seidl *et al.* 2008b).

4.3.2 Objectives and methodology

This study aims at: (i) simulating the carbon sequestration in different managed forests in Italy, and for different mountain Forest Categories; and (ii) assessing how forest management can influence the carbon sequestration potentials by adopting alternative management strategies.

Selection and description of case studies

For the purposes of this work, six case studies (forest management sites) have been selected in order to represent as broad as possible the range of biophysical conditions of the mountain forest environment in Italy. The following criteria were used for selecting the case studies: (i) selected sites should implement a Forest Management Plan (FMP) (current management guide-lines and presence of large managed forest areas); and (ii) selected sites should represent different mountainous and sub-mountainous forest conditions in Italy (variability of e.g. tree species composition, soil characteristics and climate conditions, etc.). Table 26 summarizes the main characteristics of the selected sites.

Table 26: Main characteristics of selected sites.

Site	Location	FMP area (ha)	Altitude range (m a.s.l.) (or average altitude)	Annual average temperature (°C)	Annual average precipitation (mm)
Monte Limbara, Montarbu, Altopiano di Buddusò (LMB)	North-Eastern area of Sardinia region, Southern Apennines	*11,682*	600-1,300	14	900-1,000
Petronà (PET)	Sila mountain area, Calabria region, Southern Apennines	1,189.6	900-1,700	12.2	885
Veroli (VER)	Latium region, Central Apennines	4,050	800 – 1,700	10-14	1,084
Vallombrosa (VAL)	Tuscany region, Central Apennines	1,136.5	*955*	9.7	1,377
Capracotta (CAP)	Molise region, Central Apennines	1,567.3	980-1,700	9.4	1,936
Asiago (ASI)	Veneto region, South-eastern Alps	5,925	*1,000*	7	1,500-1,800

For each selected site a complete set of information on forest stands was available from the related FMPs. Forest stands information were collected in different time periods and referred to several inventory plots within the case study areas.

Framing Forest Categories and Forest Management Systems

To make data comparable at national level, inventory plots were classified by Forest Category (FC) and Forest Management System (FMS). The classification of inventory plots by FC was mainly based on the methodologies as proposed by Barbati *et al.* (2014), and Vizzarri *et al.* (2014). The following FCs were originally taken into account: (i) Montane beech forest; (ii) Mediterranean and Anatolian fir forest; (iii) Mediterranean and Anatolian black pine forest; (iv) Thermophilous pine forest; (v) Subalpine and mountainous spruce and spruce-silver fir mixed forest; and (vi) Mediterranean evergreen oak forest. For a complete description of such FCs, the reader is referred to EEA (2006).

Inventory plots were classified by FMS according to the descriptions included in each FMP. The following FMSs were identified: (i) even-aged high forest; (ii) uneven-aged high forest; and (iii) 'high-coppice' forest (ageing coppice forests whose structure can be defined as in transition to a high forest; see e.g. Stajic *et al.* 2009). Table 27 reports the number of inventory plots as classified by FC and FMS.

Table 27: Number of available inventory plots by Forest Management System and Forest Category.

	Forest Management System (FMS)			
Forest Category (FC)	Coppice forest	Even-aged high forest	Uneven-aged high forest	High-Coppice forest (originally classified as coppice forests)
Alder forest			4	
Apennine-Corsican mountainous beech forest		79	2	6 (21)
Chestnut forest	7	1		
Illyrian mountainous beech forest		17	4	
Mediterranean and Anatolian black pine forest		15		
Mediterranean and Anatolian fir forest		38		
Mediterranean evergreen oak forest	15			
Plantations of not-site-native species and self-sown exotic forest		8		
Subalpine and mountainous spruce and spruce-silver fir mixed forest			65	
Thermophilous pine forest		28		
Turkey oak, Hungarian oak and Sessile oak forest		1	3	3

Considering the low representativeness of some FCs due to the scarcity of related inventory plots (i.e. Alder forest FC), in this study we used the following FC-FMS groups: (i) Apennine-Corsican mountainous and Illyrian mountainous beech forests (here grouped into Montane beech forest FC) – even-aged and high-coppice forests; (ii) Mediterranean and Anatolian black pine forest – even-aged high forests; (iii) Mediterranean and Anatolian fir forest – even-aged high forests; (iv) Mediterranean evergreen oak forest – high-coppice forests; (v) Subalpine and mountainous spruce and spruce-silver fir mixed forest – uneven-aged high forests; and (vi) thermophilous pine forest – even-aged high forests. Table 28 reports the main characteristics of each FC-FMS group.

Table 28: Main forest stand parameters by Forest Category and dominant tree species.

Forest Category (FC)	Dominant tree species	Forest Management System	Above-ground biomass (Mg ha^{-1})	Current Annual Increment (m^3 ha^{-1} $year^{-1}$)	Basal area (m^2 ha^{-1})	Number of stems (ha^{-1})
Mediterranean and Anatolian black pine forest	*Pinus nigra* Arn., *Pinus laricio* Poiret	Even-aged high forest	414.18 (±138.70)	14.74 (±2.42)	61.22 (±16.07)	1366.58 (±1131.83)
Mediterranean and Anatolian fir forest	*Abies alba* Mill.	Even-aged high forest	362.00 (±70.19)	8.84 (±1.38)	54.10 (±9.83)	827.26 (±261.67)
Mediterranean evergreen oak forest	*Quercus ilex* L.	High-coppice forest	236.44 (±152.82)	7.97 (±1.92)	37.85 (±14.03)	3385.83 (±1946.21)
Montane beech forest	*Fagus sylvatica* L.	Even-aged high forest	421.65 (±168.75)	9.07 (±1.82)	61.37 (±49.45)	1269.25 (±1074.75)
Montane beech forest	*Fagus sylvatica* L.	High-coppice forest	258.47 (±120.91)	9.57 (±1.65)	49.74 (±36.41)	1295.38 (±1049.12)
Subalpine and mountainous spruce and spruce-silver fir mixed forest	*Picea excelsa* Link., *Abies alba* Mill.	Uneven-aged high forest	175.08 (±45.64)	8.22 (±0.49)	35.09 (±8.79)	503.69 (±212.28)
Thermophilous pine forest	*Pinus pinea* L., *Pinus pinaster* Ait., *Pinus halepensis* Mill.	Even-aged high forest	118.61 (±78.10)	5.45 (±0.43)	26.13 (±14.13)	1318.19 (±1377.81)

CO2FIX model implementation

In last decade, CO2FIX model has been mainly used to: (i) estimate the carbon sequestration potential for a range of forest types in Europe (Nabuurs and Schelhaas 2002); (ii) quantify the effects of forest management on forest carbon stocks (e.g. Kaipainen *et al.* 2004, de Jong *et al.* 2007, Kaul *et al.* 2010); and (iii) quantify carbon contents in soil (Lemma *et al.* 2007) and in wood products (Profft *et al.* 2009). Recently, CO2FIX model has been also implemented in regional (e.g. Fiorese and Guariso 2013, Ajit *et al.* 2013) and in national studies (e.g. Fang *et al.* 2013). Therefore, since the first application of CO2FIX approach (Nabuurs and Mohren 1995), the model has been further improved by: (i) including the carbon and financial accounting modules (Groen *et al.* 2006); and (ii) quantifying model uncertainties (Nabuurs *et al.* 2008). Generally, CO2FIX model is able to simulate various management scenarios and to estimate differences in carbon dynamics associated to different management scenarios (see e.g.

Masera *et al.* 2003). Moreover, CO2FIX model can be used to accurately estimate changes in stem and total above-ground tree carbon stocks in woodlots (see e.g. Kaonga and Bayliss-Smith 2012).

In this study the CO2FIX model v.3.1 (Schelhaas *et al.* 2004) is used to simulate the C stocks in above-ground biomass over the time. CO2FIX v. 3.1 is a stand-level simulation model that quantifies the C stocks and fluxes in and between the different biomass compartments, such as stems, coarse branches, foliage, and roots. The model simulates the biomass growth (i.e. the stand development over the time) in balance with turnover, mortality, and harvesting rates for each biomass compartment. This latest version (v 3.1) includes also six modules, such as biomass, soil, wood products, bioenergy, financial, and carbon accounting. Figure 28 provides an overview of the model.

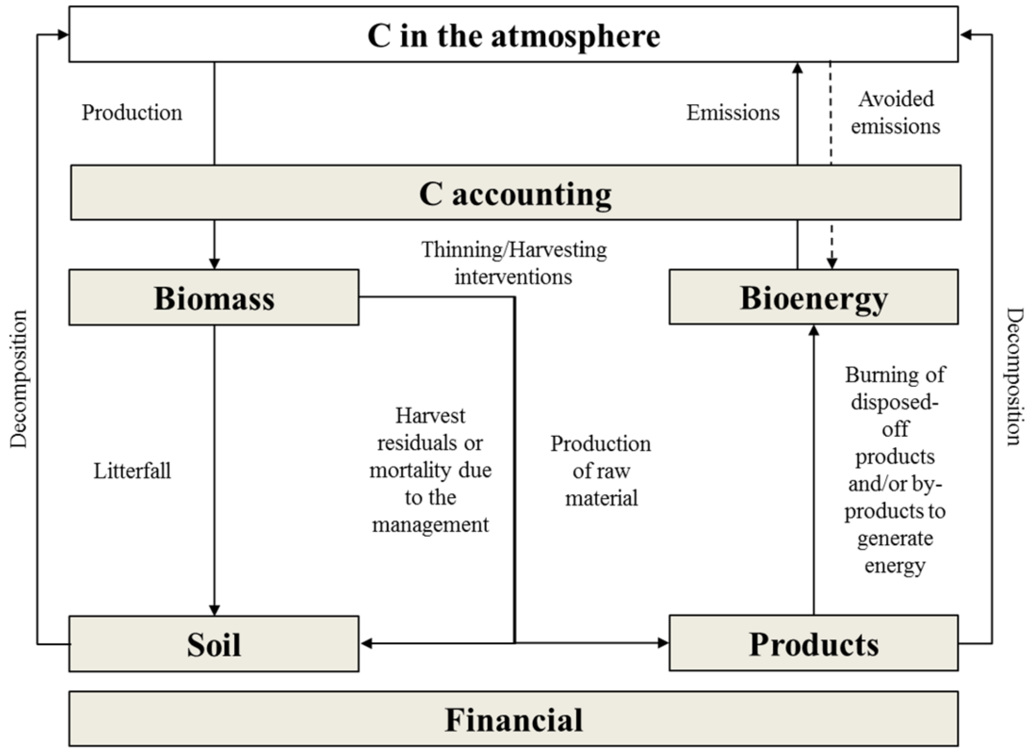

Figure 28: Flowchart reporting carbon fluxes/processes (arrows) and stocks (grey boxes) in a forest ecosystem, and as simulated by CO2FIX model (Schelhaas et al. 2004).

In order to simulate the above-ground biomass growth and subsequently the C stocks over the time, the CO2FIX model requires the following basic parameters to be set up: (i) the amount of biomass per compartment (Mg dry matter [DM] ha^{-1}); (ii) the biomass carbon content (Mg C Mg DM^{-1}); (iii) the current annual increment (CAI for the stems compartment, m^3 ha^{-1} $year^{-1}$); (iv) the mortality rates (percentage $year^{-1}$); and (v) the turnover rates (for the coarse branches, foliage and roots compartments; $year^{-1}$). Table 29 reports the description of the input data that were used for all FCs and FMSs as model parameters in this study.

Table 29: Summary description of the most important input data as used for CO2FIX model parameterization.

Biomass compartment	Input type	Input value	Main reference	Description
All	Amount of biomass (Mg DM ha^{-1})	DIM	Local FMPs	For each inventory plot, the available volume of growing stock (m^3 ha^{-1}) was converted into the volume of the above-ground biomass by adopting the following equation (Federici *et al.* 2008): $B_{AG} = GS \cdot BEF \cdot WBD$ where: B_{AG} is the volume of the above-ground woody tree biomass (Mg DM ha^{-1}); GS is the volume of growing stock (m^3 ha^{-1}); BEF is the biomass expansion factor that expands the growing stock volume to the volume of the above-ground woody biomass; WBD is the wood basic density (Mg DM m^{-3}). BEF values for the main FCs and FMSs in Italy are available from Federici *et al.* (2008).
Stems	CAI (m^3 ha^{-1} year^{-1})	DIM	Local FMPs	CAI available for all inventory plots.
Stems	Mortality rate (% DM)	0.005	Pietsch *et al.* (2005)	Not available local data.
Foliage	Turnover rate (% DM year^{-1})	Values by tree species	Pietsch *et al.* (2005)	Not available local data.
Coarse branches and roots	Turnover rate (% DM year^{-1})	0.025	None	Considering the absence of local data and/or adaptable scientific references, and taking into account that the turnover rate mainly influences the soil C stock (and not the above-ground biomass as in this study), it was arbitrarily chosen.

For the purposes of this work, the simulations concern the C stock trend in above-ground biomass (in terms of Mg C ha^{-1}) along a 300-year period. In CO2FIX model, simulations can be differentiated in terms of frequency and intensity of thinning/harvesting interventions (see e.g. Masera *et al.* 2003). The frequency is defined by setting the age at which a specific intervention takes place, whilst the intensity is defined by the utilization rate (ratio between the amount of thinned/harvested biomass and the total amount of biomass of the stand at a specific time-step).

For each FC-FMS group (see Table 27), a panel of experts defined type, frequency and intensity of intervention. In this study, simulations were grouped as follows: "Regular Management (RM)" and "Alternative Management (AM)". For RM, intensity and frequency of interventions were defined by FC and FMS according to the most representative management guide-lines and prescriptions as available at national level, and generally described in FMPs for all case-studies. By other hand, for AM intensity and frequency of interventions were defined according to the management guide-lines and prescriptions as reported in FMPs for Montane beech forests in two case-studies, CAP (high-coppice forest) and ASI (uneven-aged high forest). Appendix 3 reports the details about management interventions as defined and used for initializing the CO2FIX model simulations.

4.3.3 Results

Simulation of carbon stock with regular management regime

Figure 29 shows the simulation of C stock in above-ground biomass for the Mediterranean-Anatolian black pine, even-aged forests.

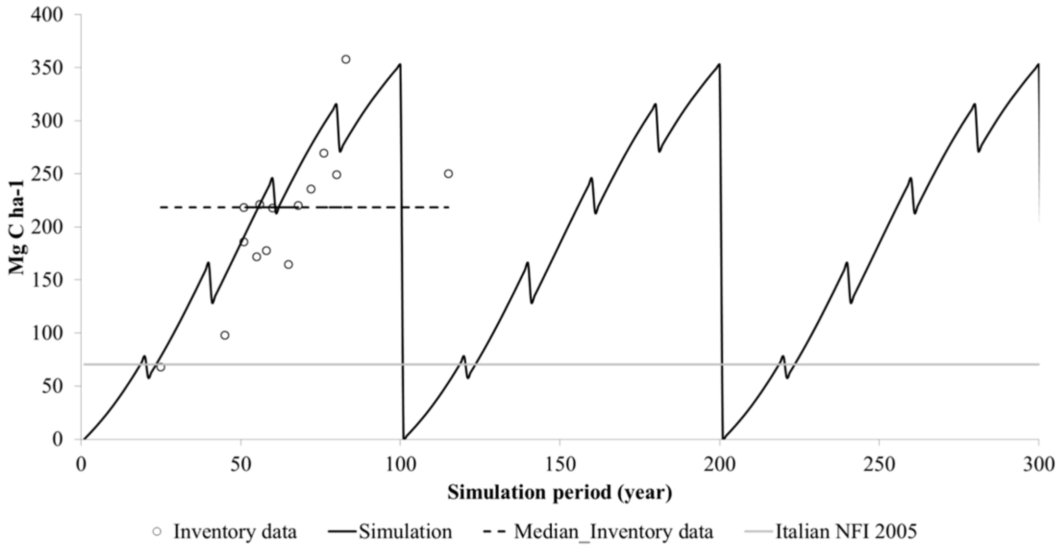

Figure 29: 300-year simulation of C stock in above-ground biomass for Mediterranean and Anatolian black pine forest managed with clearcutting system.

In the case of black pine even-aged forests managed with a clearcutting system, C stock in above-ground biomass increases up to 352.8 Mg C ha^{-1} at the end of rotation period (100-year step). If compared with the median value (218.4 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 55, by differing of approximately 135 Mg C ha^{-1} before final felling. 20-year step thinnings averagely remove 56 Mg C ha^{-1} during the entire rotation period. Considering the cumulative C stock for the entire simulation period (300 years), RM contributes to increase the C stock in above-ground biomass as follows: (i) up to 61 Mg C ha^{-1} at the end of the first rotation (100 years); (ii) up to 122 Mg C ha^{-1} at the end of the second rotation (200 years); and (iii) up to 183 Mg C ha^{-1} at the end of simulation period (300 years). The difference between the highest simulated C stock value and the reference value from the Italian NFI is approximately +282.4 Mg C ha^{-1}.

Figure 30 shows the simulation of C stock in above-ground biomass for the Mediterranean-Anatolian silver fir, even-aged forests.

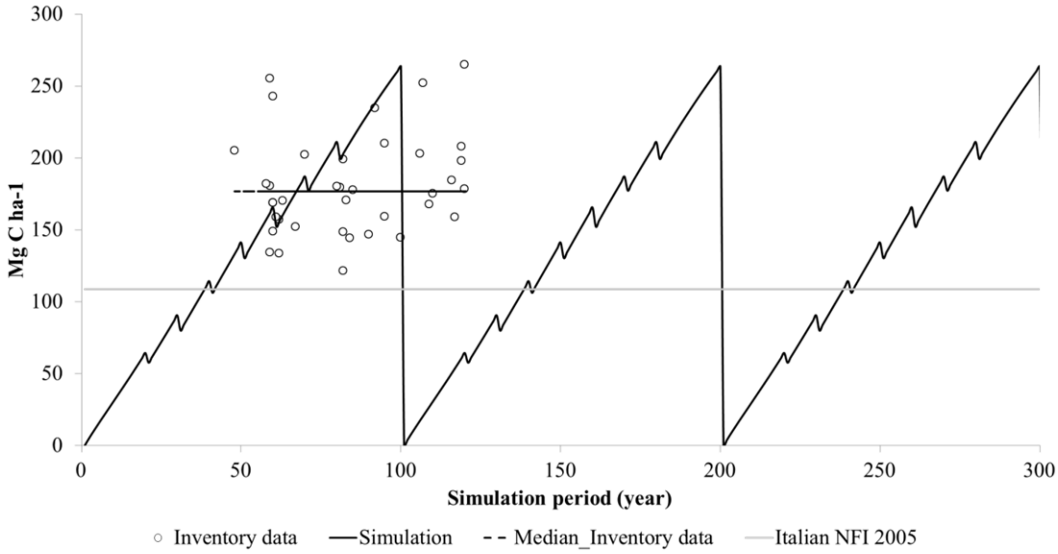

Figure 30: 300-year simulation of C stock in above-ground biomass for Mediterranean-Anatolian silver fir even-aged forest managed with clearcutting system.

In the case of silver fir even-aged forests managed by clearcutting system, C stock in above-ground biomass increases up to 263.5 Mg C ha^{-1} at the end of rotation period (100-year step). If compared with the median value (176.8 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 68, by differing of approximately 86.7 Mg C ha^{-1} before final felling. 10-year step thinnings (excepting than for both the first and last 20 years) averagely remove 10 Mg C ha^{-1} during the entire rotation period. Considering the cumulative C stock for the entire simulation period (300 years), RM contributes to increase the C stock in above-ground biomass as follows: (i) up to 45 Mg C ha^{-1} at the end of the first rotation (100 years); (ii) up to 89 Mg C ha^{-1} at the end of the second rotation (200 years); and (iii) up to 134 Mg C ha^{-1} at the end of simulation period (300 years). The difference between the highest simulated C stock value and the reference value from the Italian NFI is approximately +130 Mg C ha^{-1}.

Figure 31 shows the simulation of C stock in above-ground biomass for the montane beech, even-aged forests.

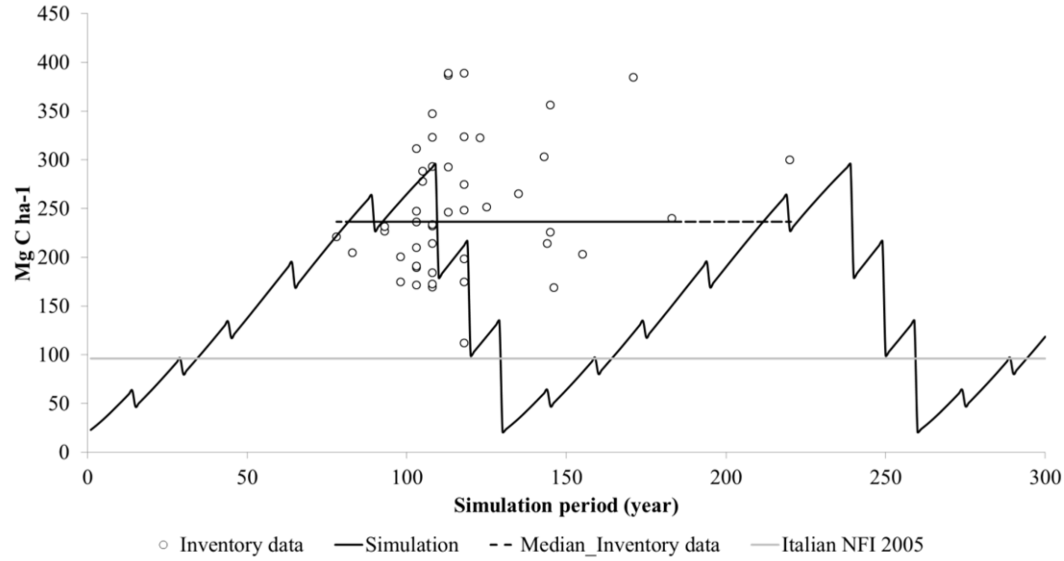

Figure 31: 300-year simulation of C stock in above-ground biomass for montane beech forest managed with shelterwood system.

In the case of montane beech, even-aged forests managed by shelterwood system, C stock in above-ground biomass increases up to 295.5 Mg C ha^{-1} before the seed cut (110 years). If compared with the median value (236.5 Mg C ha^{-1}) from inventory plots, simulated C stock is higher in the 82-89 year period (by differing of approximately 27.2 Mg C ha^{-1}, before the last thinning), and in the 93-109 year period (by differing of approximately 59 Mg C ha^{-1}, before the seed cut). However, the highest C stock value is lower than the highest one from inventory plots (389 Mg C ha^{-1}). Along the rotation period, thinnings averagely remove 22.4 Mg C ha^{-1}. Considering the cumulative C stock for the entire simulation period (300 years), RM contributes to increase the C stock in above-ground biomass as follows: (i) up to 67.4 Mg C ha^{-1} at the end of first rotation period (final cut, 130 years); and (ii) up to 135 Mg C ha^{-1} at the end of second rotation period (final cut, 260 years). The difference between the highest simulated C stock value and the reference value from Italian NFI is approximately +292.8 Mg C ha^{-1}.

Figure 32 shows the simulation of C stock in above-ground biomass for the montane beech, high-coppice forests.

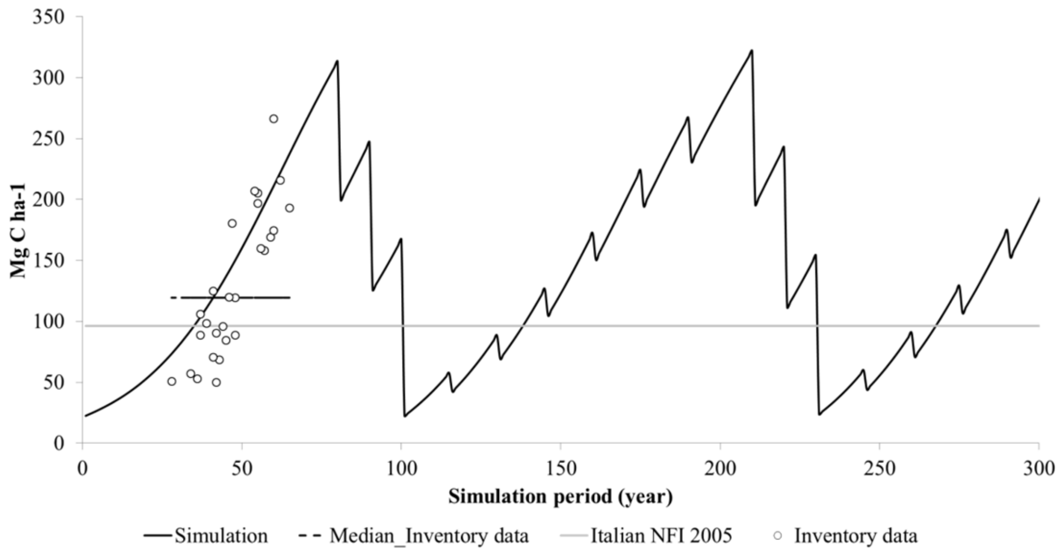

Figure 32: 300-year simulation of C stock in above-ground biomass for montane beech, high-coppice forest, managed by conversion of ageing coppice stand and then with shelterwood system.

In the case of montane beech, high-coppice forests managed by selection system (conversion into high forest and then shelterwood system), simulated C stock in above-ground biomass increases up to 313.1 Mg C ha^{-1} after natural development of the ageing coppice stand (without intermediate thinnings) and before starting the conversion process (year 80). After that, an higher level is reached at the end of the next rotation period (321.8 Mg C ha^{-1}; 210 years), when the new high forest stand is ready for the seed cut. If compared with the median value (119.4 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 42, by differing of approximately 193.7 Mg C ha^{-1} until the secondary cut (at year 90). Between year 100 and year 200, thinnings averagely remove 23.7 Mg C ha^{-1}. Considering the cumulative C stock for the entire period (300 years), RM contributes to increase the C stock in above-ground biomass as follows: (i) up to 30.5 Mg C ha^{-1} at year 80, and before starting the conversion process; and (ii) up to 97.6 Mg C ha^{-1} at year 210, which represent the end of rotation period for the new high forest stand. The difference between the highest simulated C stock value and the reference value from Italian NFI is approximately +225.6 Mg C ha^{-1}.

Figure 33 shows the simulation of C stock in above-ground biomass for the Mediterranean evergreen oak, high-coppice forests.

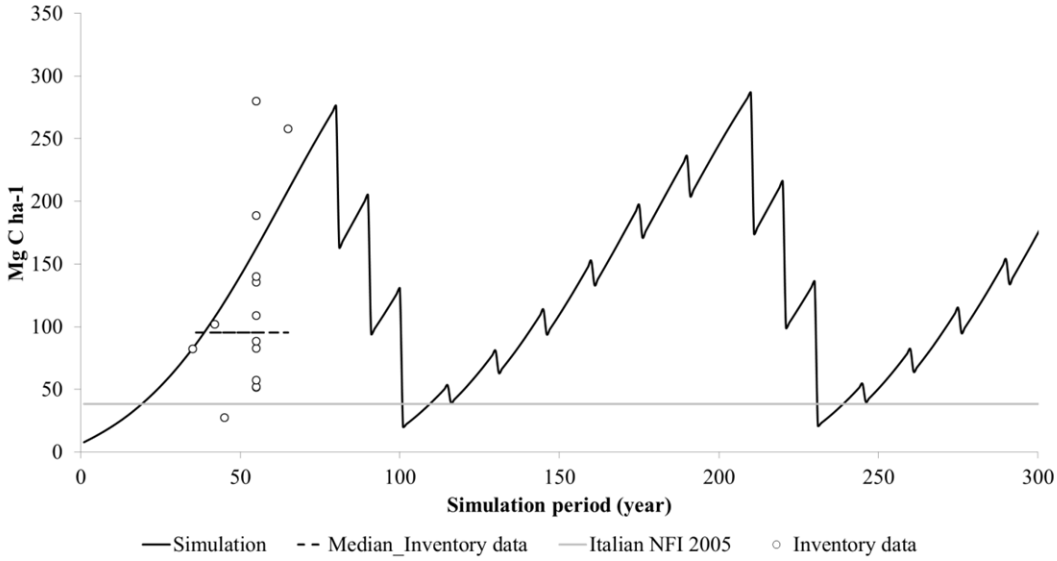

Figure 33: 300-year simulation of C stock in above-ground biomass for Mediterranean evergreen oak, high-coppice forest managed by conversion of ageing coppice stand and then with shelterwood system.

In the case of Mediterranean evergreen oak, high-coppice forests managed by selection system (conversion into high forest through natural development and shelterwood system), C stock in above-ground biomass increases up to 276 Mg C ha^{-1} after natural development of the ageing coppice stand (without intermediate thinnings) and before starting the conversion process (year 80). After that, an higher level is reached at the end of the next rotation period (286.6 Mg C ha^{-1}; 210 years), when the new high forest stand is ready for the seed cut. If compared with the median value (95.4 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 39, by differing of approximately 180.6 Mg C ha^{-1} until the seed cut (at year 80). Between year 100 and year 200, thinnings averagely remove 21.2 Mg C ha^{-1}. Considering the cumulative C stock for the entire period (300 years), RM contributes to increase the C stock in above-ground biomass as follows: (i) up to 31.3 Mg C ha^{-1} at year 80, and before starting the conversion process; and (ii) up to 91.2 Mg C ha^{-1} at year 210, which represent the end of rotation period for the new high forest stand. The difference between the highest simulated C stock value and the reference value from Italian NFI is approximately +248.2 Mg C ha^{-1}.

Figure 34 shows the simulation of C stock in above-ground biomass for the Subalpine and mountainous spruce-silver fir mixed, uneven-aged forests.

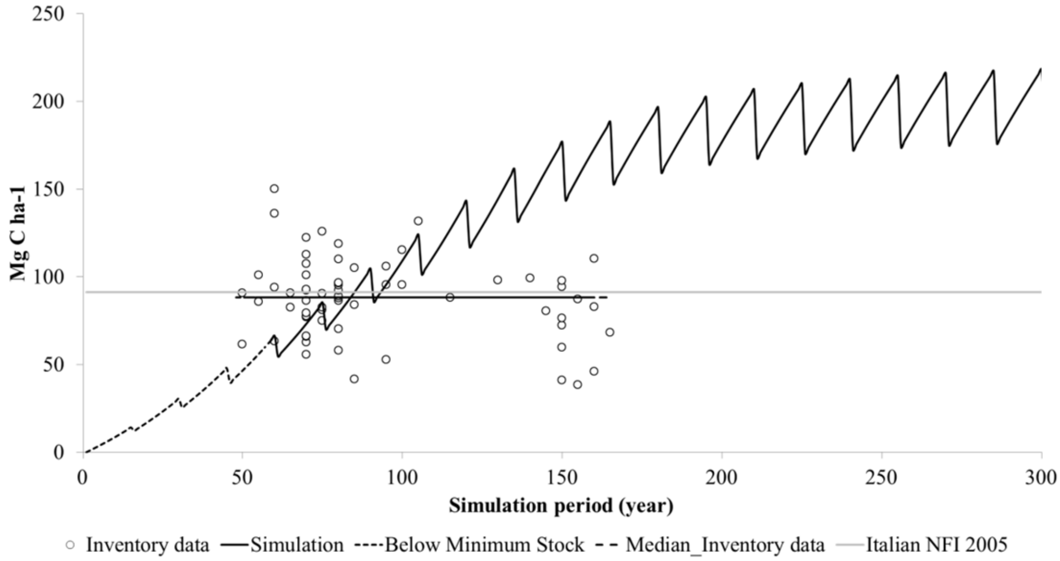

Figure 34: 300-year simulation of C stock in above-ground biomass for Subalpine and mountainous spruce-silver fir mixed, uneven-aged forest managed with selection system.

In the case of subalpine and mountainous spruce-silver fir mixed, uneven-aged forests managed by selection system (repeated thinnings), C stock in above-ground biomass increases up to 218.2 Mg C ha^{-1} by the end of simulation period. C stock value remains stable after 150 years, and ranges between 140 (lower level) and 220 Mg C ha^{-1} (upper level). If compared with the median value (88.3 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 84, by differing of about 88.5 Mg C ha^{-1} at the middle of simulation period (150 years). Over the entire simulation period (300 years), intermediate thinnings averagely remove 28.2 Mg C ha^{-1}. Considering the cumulative C stock for the entire period (300 years), RM contributes to increase the C stock in above-ground biomass as follows: (i) up to 16.9 Mg C ha^{-1} at year 100; (ii) up to 68.6 Mg C ha^{-1} at year 200; and (iii) up to 133.3 Mg C ha^{-1} at year 300. The difference between the highest simulated C stock value and the reference value from Italian NFI is approximately +126.9 Mg C ha^{-1}.

Figure 35 shows the simulation of C stock in above-ground biomass for the thermophilous pine, even-aged forests.

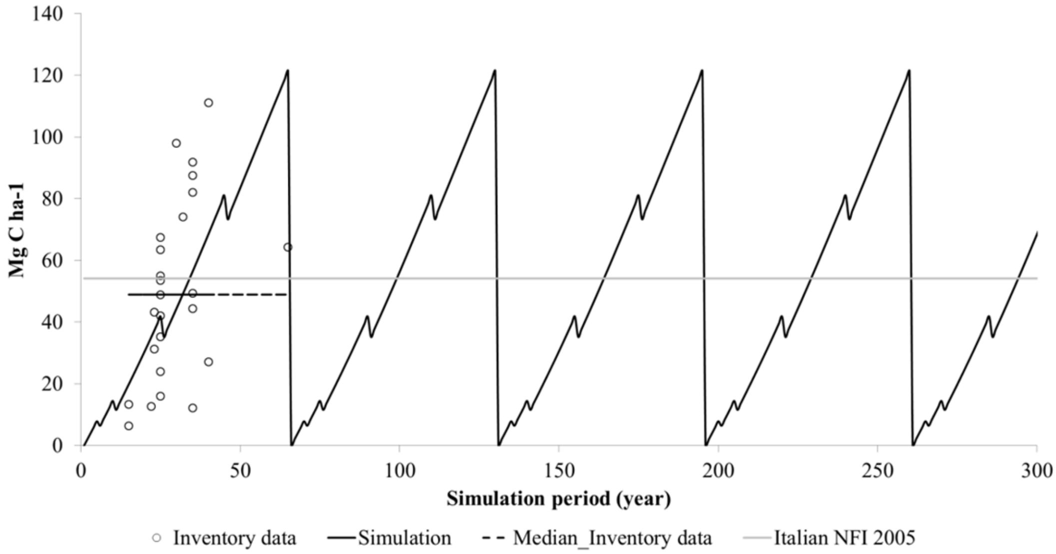

Figure 35: 300-year simulation of C stock in above-ground biomass for Thermophilous pine, even-aged forest managed with clearcutting system.

In the case of thermophilous pine, even-aged forest managed by clearcutting system, C stock in above-ground biomass increases up to 121.4 Mg C ha^{-1} by the end of rotation period (year 65). If compared with the median value (48.9 Mg C ha^{-1}) from inventory plots, simulated carbon stock is always higher after year 32, by differing of approximately 72 Mg C ha^{-1}. Over the rotation period, repetead thinnings averagely remove 4.6 Mg C ha^{-1}. Considering the cumulative C stock for the entire period (300 years), RM contributes to increase C stock in above-ground biomass as follows: (i) up to 11.9 Mg C ha^{-1} at the end of first rotation period (year 65); (ii) up to 23.8 Mg C ha^{-1} at the end of second rotation period (year 130); (iii) up to 35.7 Mg C ha^{-1} at the end of third rotation period (year 195); (iv) up to 47.6 Mg C ha^{-1} at the end of fourth rotation period (year 260); and (v) up to 51.7 Mg C ha^{-1} at the end of simulation period (year 300). The difference between the highest simulated C stock value and the reference value from Italian NFI is approximately +67.2 Mg C ha^{-1}.

Simulation of carbon stock with alternative management regime

Montane beech forests in Capracotta

Figure 36 shows the simulation of C stock in above-ground biomass for the montane beech, high-coppice forests in CAP case study.

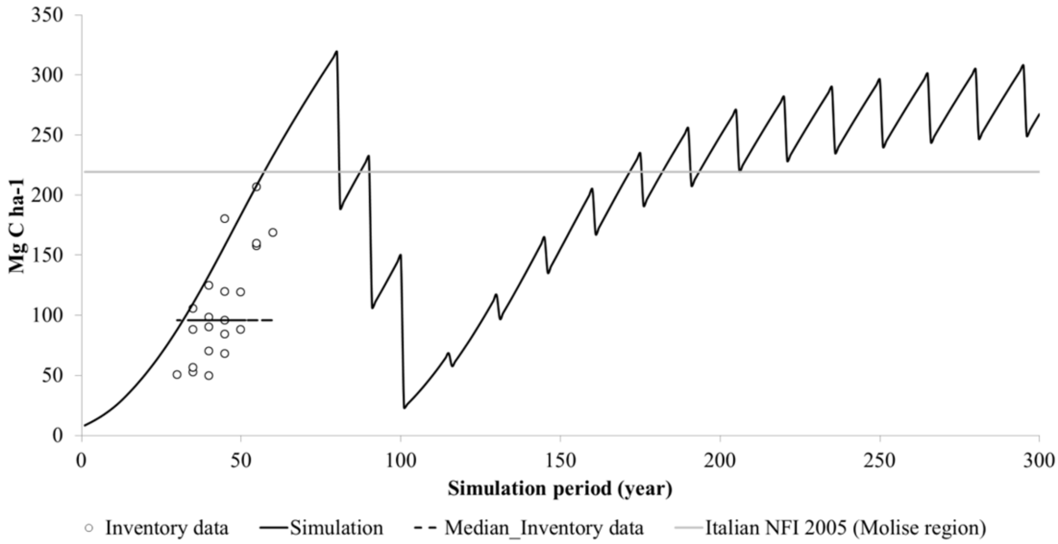

Figure 36: 300-year simulation of C stock in above-ground biomass for Montane beech, high-coppice forest in CAP managed by conversion of ageing coppice stand and then with selection system.

In the case of montane beech, high-coppice forests managed by selection system (conversion to high forest) in CAP case study, simulated C stock in above-ground biomass increases up to 319.1 Mg C ha^{-1} after natural development of the ageing coppice stand (without intermediate thinnings) and before starting the conversion process (year 80). After that, another high level is reached nearly to the end of simulation period (307.8 Mg C ha^{-1}; year 295). If compared with the median value (95.9 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 32, by differing of approximately 223.2 Mg C ha^{-1} until starting the conversion process. In the 100-300 year period, thinnings averagely remove 44.2 Mg C ha^{-1}. Considering the cumulative C stock for the entire period (300 years), AM contributes to increase the C stock in above-ground biomass as follows: (i) up to 38.9 Mg C ha^{-1} at year 80, and before starting the conversion process; and (ii) up to 190.3 Mg C ha^{-1} at the end of simulation period (year 300). The difference between the highest simulated C stock value and the reference value from Italian NFI (for Molise region) is approximately +91.5 Mg C ha^{-1}.

Montane beech forests in Asiago

Figure 37 shows the simulation of C stock in above-ground biomass for the montane beech forests in ASI case study.

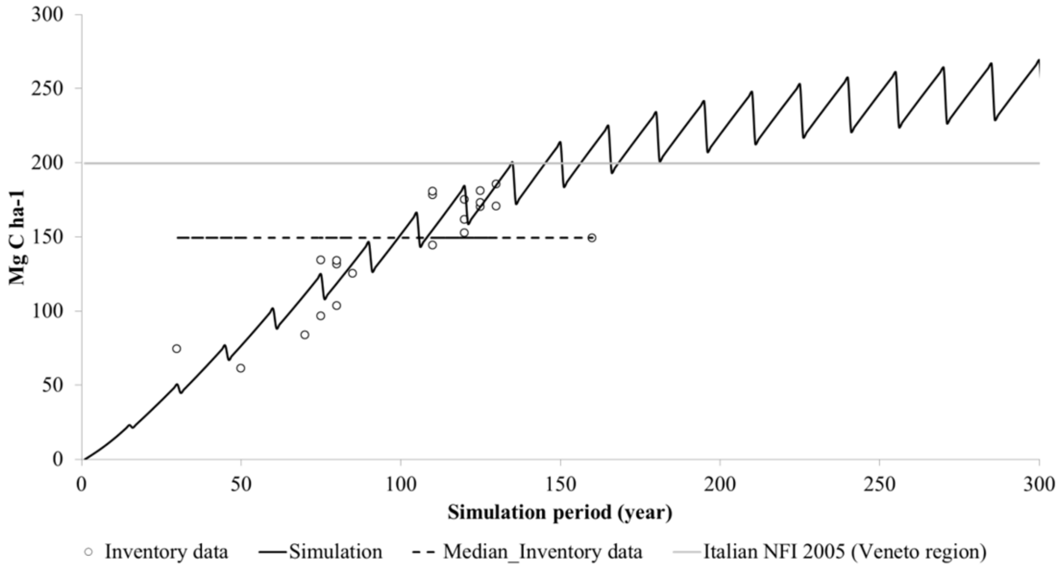

Figure 37: 300-year simulation of C stock in above-ground biomass for Montane beech, uneven-aged forest in ASI managed with selection system.

In the case of montane beech forests managed by selection system in ASI case study, simulated C stock in above-ground biomass increases up to 269 Mg C ha^{-1} by the end of simulation period. C stock value remains stable after 200 years, and ranges between 220 (lower level) and 260 Mg C ha^{-1} (upper level). If compared with the median value (149.5 Mg C ha^{-1}) from inventory plots, simulated C stock is always higher after year 100, by differing of about 63.9 Mg C ha^{-1} at the middle of simulation period (150 years). Over the entire simulation period (300 years), intermediate thinnings averagely remove 26.2 Mg C ha^{-1}. Considering the cumulative C stock for the entire period (300 years), AM contributes to increase the C stock in above-ground biomass as follows: (i) up to 25.9 Mg C ha^{-1} at year 100; (ii) up to 90.9 Mg C ha^{-1} at year 200; and (iii) up to 171.4 Mg C ha^{-1} at year 300. The difference between the highest simulated C stock value and the reference value from Italian NFI (for Veneto region) is approximately +69.3 Mg C ha^{-1}.

4.3.4 Discussion and conclusions

Results showed the C sequestration potential of several FCs in mountain environments in Italy, in terms of simulated C stock over 300-year simulation period, according to different management systems and strategies (i.e. intensity and frequency of forestry interventions). According to the results, C stock (and subsequently C sequestration potential) is generally influenced over the time by the following main factors: (i) site- and species-specific parameters about e.g. CAI, turnover and mortality rates, stand ages, etc.; and (ii) forest management, in terms of e.g. rotation period, harvesting intensity (thinning and final felling), frequency of interventions, etc. Such influencing factors are hereinafter discussed.

Carbon stock variability among Forest Categories

Simulated C stocks in above-ground biomass range between 121.4 Mg C ha^{-1} in the case of Thermophilous pine, even-aged forests (lowest value; Figure 35) and 352.8 Mg C ha^{-1} in the case of Mediterranean and Anatolian black pine, even-aged forests (highest value; Figure 29). For example, in the case of Thermophilous pine forests, Gasparini and Di Cosmo (2015) reported similar values at national scale (133 Mg C ha^{-1}), as well as Ruiz-Peinado *et al.* (2013) for Spain (between 91 and 137.5 Mg C ha^{-1}, according to the management intensity). For broadleaved forests, simulated C stocks in above-ground biomass range between 286 Mg C ha^{-1} in the case of Mediterranean evergreen oak, high-coppice forests (lowest value; Figure 33) to 321 Mg C ha^{-1} in the case of Montane beech, high-coppice forests (highest value; Figure 32). With regards to Montane beech forests (both even-aged and high-coppice), simulated C stock in above-ground biomass is higher than that obtained in previous studies in Italy (123 Mg C ha^{-1}; Bayat *et al.* 2012) , in Spain (129 Mg C ha^{-1}; Merino *et al.* 2007), and in Germany (120 – 160 Mg C ha^{-1}; Joosten *et al.* 2004). These differences are strictly related to the simulated RM, especially in the case of converting ageing coppice stands to high-forest and then adopting a shelterwood system. Therefore, cited studies refer to specific sites, and do not capture the whole C stock trend from a broader perspective (i.e. national scale), as in this study.

Excepting than for Thermophilous pine forests, simulated C stock values for coniferous forests are generally higher in comparison with those simulated for broadleaved forests. This depends on a higher C accumulation in coarse branches in the case of conifers. Generally, C stock values from simulation are consistent with those collected from inventory plots, excepting than for Montane beech, even-aged high forests. In fact, for this FC, the highest simulated C stock value is lower than the highest value from inventory data (see Figure 31). This depends on high data variability among beech forests, in terms of biophysical and climatic characteristics (e.g. between subalpine and Apennine beech forests; see Table 28). Moreover, the C stock differences between simulation and inventory data for beech forests may depend on the intensity and frequency of interventions (i.e. shelterwood system; RM). As also described by Nocentini *et al.* (2009), practical forest management in beech stands was and still is referred to selection systems, which are based on specific needs of the forest owners (in most cases, small private forest owners) and peculiar stand characteristics among sites. Even in the case of Subalpine and mountainous spruce-silver fir mixed forests, simulation does not fit the C stock values from inventory data, especially with regards to ageing plots (around 150 years). This gap may depend on the hidden methodology that was used to assign ages to each stand in FMPs. In most cases, age was assigned to dominant trees to each forest parcel, and as a consequence was not representative of the real stand structures by age class. This also explains the complete absence of C stocks about younger age classes from inventory data. However, age-biomass correlation has not to be considered when treating uneven-aged forests, as in the case of Subalpine and mountainous spruce-silver fir mixed ones.

As partly mentioned before, simulated C stock trends strictly depend on the accumulation of biomass over the time. Subsequently, the C stock variability among FCs depends on the growing capacity of the related stands (i.e. CAI). For coniferous forests, a lower C stock value in the case of Thermophilous pine forests is regulated by a lower average CAI in comparison with that obtained for Mediterranean and Anatolian black pine forests (5.45 and 14.7 m^3 ha^{-1} $year^{-1}$, respectively). Considering the input data, Thermophilous pine forests refer to the LMB case study, which is generally representative of more adverse climate conditions (i.e. drought possibility, scarce precipitations, etc.) and degraded soils in comparison with the more favorable climate conditions and fertile soils in VER and PET case studies (for Mediterranean and Anatolian black pine forests). By other hand, although both FCs are managed by adopting a clearcutting system, a shorter rotation period for Thermophilous pine forests in comparison with that for Mediterranean and Anatolian black pine forests (65 and 100 years, respectively) strongly reduces the stand growing capacity over the time. The same considerations can be reported for broadleaved forests, as in the cases of Montane beech forests and Mediterranean evergreen oak forests. Although both FCs are managed by adopting a conversion to high-forest and then a shelterwood system, Mediterranean evergreen oak forests have a lower average CAI in comparison with Montane beech forests (7.6 and 9.2 m^3 ha^{-1} $year^{-1}$, respectively). As for the coniferous forests, climate conditions and soil fertility are two of the most important factors influencing above-ground biomass growth and subsequently C sequestration over the time.

In any case, simulated C stock values are always higher than those estimated by the Italian NFI, and range between +67.2 Mg C ha^{-1} in the case of Thermophilous pine, even-aged forests and +292.8 Mg C ha^{-1} in the case of Montane beech, even-aged forests (see Figure 35 and Figure 31). The higher gap showed by broadleaved forests is mainly due to the fact that Italian NFI estimates do not differentiate forest stands according to different management systems (high- and coppice forests) and ages (both even-aged and uneven-aged stands) (Gasparini and Tabacchi 2011). Even for the coniferous stands there is no differentiation only between ages (even- and uneven-aged forests) (Gasparini and Tabacchi 2011). By other hand, the gap between simulated C stock values and Italian NFI estimates reduces while downscaling at case study level, as Montane beech forests in CAP and ASI case studies (see Figure 36 and Figure 37, respectively).

Carbon sequestration potential variability with different Management Approaches

Among investigated FCs, Montane beech, high-coppice forests and Mediterranean evergreen oak, high coppice forests are found to have the greatest potential to sequester C in above-ground biomass over the long run (until the end of simulation period) (see Figure 32 and Figure 33, respectively). By the contrary, Thermophilous pine, even-aged forests and Mediterranean and Anatolian silver fir, even-aged forests are found to have the lowest potential to sequester C in above-ground biomass (see Figure 35 and Figure 31, respectively).The C sequestration potential variability between FCs is mainly driven by the differences between management approaches (i.e. RM; see also e.g. Jandl *et al.* 2007; Bravo *et al.* 2008). In this way, the balance between CAI and the intensity/frequency of interventions plays a key role in increasing the C stock, especially over the long run. For example, although Mediterranean evergreen oak, high-coppice forests are found to have a low average CAI, the conversion to high-forest by allowing a natural stand development during the first period, and then adopting the shelterwood system, ensures the forest stand capability to accumulate C (up to 91.2 Mg C ha^{-1} at the end of rotation period). By other hand, in the case of Thermophilous pine, high-coppice forests, adopted RM (clearcutting system with a rotation period of 65 years) strongly reduces the residence time to accumulate C in above-ground biomass. Thinning regime also influences the C sequestration potential, in terms of their frequency and intensity (see e.g. Pussinen *et al.* 2009 at European scale). Indeed, thinning regime affects the stand capacity to accumulate C in relative short time periods (see e.g. Bradford *et al.* 2013). For example, simulations for Montane beech, high-coppice forests and Mediterranean evergreen oak forests showed less-impacting thinnings (23.7 and 21.2 Mg C ha^{-1} removed, respectively, along 90 years for both) in comparison with much more impacting thinnings as simulated for Thermophilous pine forests (56 Mg C ha^{-1} along 65 years).

Through contrasting RM with AM and downscaling the them from national to local scale, results demonstrate important effects of different management approaches with regards to C stock and C sequestration potential. In the case of Montane beech, high-coppice forests (national scale and CAP), the shelterwood system after converting the ageing coppice stand (RM) facilitates the increasing in C stock up to 321.8 Mg C ha^{-1} at the end of the rotation period (and before the seed cut). This value is higher rather than the C stock value obtained by adopting a selection system after the conversion to high-forest (AM). This is due to the intensity and frequency of thinnings, which are more impacting with AM (AM=44.2 Mg C ha^{-1} averagely removed; RM= 23.7 Mg C ha^{-1} averagely removed). However, C sequestration potential is relatively higher with AM at the end of simulation/rotation period (AM=190.3 Mg C ha^{-1} at year 300; RM=97.6 Mg C ha^{-1} at year 210). Similarly in the case of Montane beech, even-aged forests (national scale and ASI), the shelterwood system (RM) increases the C stock up to 295.5 Mg C ha^{-1} at the end of the rotation period (and before the seed cut). This value is higher than the C stock value obtained by adopting a selection system (AM). Even in this case, this depends on the intensity and frequency of thinnings, which are more impacting with AM (AM=26.2 Mg C ha^{-1} averagely removed; RM=22.4 Mg C ha^{-1} averagely removed). Nevertheless, C sequestration potential is relatively higher with AM at the end of simulation/rotation period (AM=171.4 Mg C ha^{-1} at year 300; RM=135 Mg C ha^{-1} at year 260). Results globally demonstrated that RM, by increasing the intensity (in terms of biomass removed) and frequency of interventions (thinnings and final felling) has a negative impact on C sequestration potentials over the long run. On the contrary, AM through selective thinnings (i.e. continuous cover forestry) is found to have lower C stock values, especially at certain time steps.

Sensitiveness of the model

CO2FIX demonstrated to be a suitable model to simulated biomass-based development of investigated forests, and as a consequence to calculate the related C stock in above-ground biomass for different FCs in Italy. Considering that CO2FIX model performs a separate calculation for three above-ground biomass compartments, such as stems, foliage and coarse branches, results demonstrated the biomass accumulation to be very sensitive to little variations of such parameters. In this sense, the robustness of input data, especially with regards to the biomass compartment-relative growth (CAI) and C allocation, has a fundamental role to better parameterize the model and subsequently to have simulations being consistent with inventory plots. A general lack of site- and species-specific information about turnover and mortality rates, especially for foliage and coarse branches compartments, made very difficult the model parameterization and subsequently the interpretation of results. Moreover, the methodology chosen for calculating above-ground biomass from standing volume (Federici *et al.* 2008) is affected by a relative uncertainty, which has an influence on C accumulation over the time. Considering that a model validation was not was not carried out in this study, the comparison between simulated C stocks and the associated values from inventory data revealed a certain consistence between model outputs and available standing C stocks. This was rather different with regards to Montane beech forests and Subalpine and montane spruce-silver fir forests, for which the high data variability and the adopted forest management system affected the model fitting.

Van der Voet (Nabuurs and Mohren 1993) carried out an uncertainty analysis of the model CO2FIX model for the Norway spruce FC in central Europe. For the 32 independent inputs to the model, he found that for the total carbon stock, the average amounted to 316 Mg C ha^{-1}, whereas the 95% confidence interval ranged from 254 to 403 Mg C ha^{-1} which was found to be reasonable. In addition, Nabuurs *et al.* (2008) used the CO2FIX model to calculate the sensitivity and the uncertainty analyses while analyzing the carbon sequestration in two sites, such as a Norway spruce forest in Western Europe, and a complex tropical forest in northern Costa Rica. They found that the spruce case (high data availability) shows an uncertainty range (95% confidence interval) of 100 Mg C ha^{-1} for the forest and soils C stock (on an average of 207 Mg C ha^{-1}) while in the secondary tropical forest case (low data availability) the uncertainty range is 195 Mg C ha^{-1} (on an average of 113 Mg C ha^{-1}) (Nabuurs *et al.* 2008).

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5

Future challenges to implement the "resilience thinking" in forest management

This chapter summarizes the main research findings and future-oriented perspectives on how to implement the "resilience thinking" in forest management. At first, the role of forest management in improving ecosystem resilience is described. Then, the importance of understanding the relationships between forest ecosystem functioning, biodiversity and ecosystem services provision for improving forest ecosystem resilience is further explained through providing management guide-lines both at stand and landscape level. Contextually, the substantial role of traditional forest management and local communities engagement is outlined. Finally, a list of key messages towards the "resilience thinking" is provided.

5.1 Forest ecosystem complexity, resilience capacity, and management: lessons learned

The word 'resilience' encompasses the following three attributes of a forest ecosystem: (i) the ability to cope with stress; (ii) the capacity to recover from the effects of disturbance; and (iii) the capability to adapt to stress and change (see Chapter 2). As previously described (see Chapter 1), external impacts on forest ecosystem resilience mainly refer to (i) climate change-induced modifications (e.g. forest fires, drought and increased stand mortality, insects outbreaks, extreme atmospheric events, loss of site-native tree species, etc.), and anthropogenic disturbances (e.g. land use and cover change, loss and fragmentation of habitats, unsustainable management practices, soil degradation, reduction of water quality and quantity, etc.). These external changes directly impact the forest biodiversity and subsequently the delivery of important services to local communities (see e.g. Cardinale *et al.* 2012; Mace *et al.* 2012).

Within the forestry sector, understanding and analyzing the factors impacting the forests' resilience is extremely important to orient decisions towards improving the functioning of forest ecosystems and subsequently the benefits' flow to people. In this way, managing forests to improve ecosystem resilience concerns the improvement of both adaptation and stability of forest ecosystems to environmental changes from stand (sustainable forestry interventions) to landscape level (integrated and ecosystem-based approaches) (see e.g. Rist and Moen 2013). Increasing resilience of forest and trees to climate change through forest management include the following key strategies (see e.g. Braatz 2012): (i) maintaining healthy forest ecosystem for resilience (Rapport *et al.* 1998); (ii) restoring degraded forests (Lamb *et al.* 2005; Chazdon 2008); and (iii) conserving, enhancing and using biodiversity (Fischer *et al.* 2006). Definitions of ecosystem health have been closely allied with the concepts of stress ecology, which define health in terms of "system organization, resilience and vigor, as well as the absence of signs of ecosystem distress" (see e.g. Costanza *et al.* 1992).

Increasing resilience of forest ecosystems to land use change and other anthropogenic disturbances generally requires assessing and managing inherent tradeoffs between meeting immediate human needs and maintaining the capacity of ecosystems to provide goods and services in the future (DeFries *et al.* 2004). Assessing trade-offs among multiple benefits must recognize that land use provides fundamental goods and services, even while originating ecosystem degradation and long-term declines in human welfare (DeFries *et al.* 2004). Moreover, sustainable land use policies and less-impacting management strategies enhances the resilience of different land use practices (Foley *et al.* 2005). Increasing the resilience of managed landscapes requires integrated approaches particularly suitable to maintain the landscape asset as multi-functional as possible.

According to the previously reported research findings, forest management can be considered as one of the most important drivers influencing forest ecosystem resilience and functioning. Through adopting alternative management strategies, forest ecosystem services extremely vary among future-oriented target states (see Chapters 3 and 4). Accordingly, integrating 'resilience thinking' in forest management requires a whole understanding of the effects of anthropogenic disturbances (i.e. forest management) on ecosystem functioning and stability. Moreover, valorizing the role of local communities in decision-making contexts and management practices implementation is a key strategy to promote "resilience thinking" in forest management at different scales.

5.2 Understanding the relationships between forest biodiversity and ecosystem services

In many cases, the terms 'biodiversity' and 'ecosystem services' (BES) are used simultaneously, implying that they are effectively the same thing and that if ecosystem services are managed in the best way, biodiversity will be retained and *vice versa* (Mace *et al.* 2012). Cardinale *et al.* (2012) stated that biodiversity influences ecosystem functioning in the following ways: (i) biodiversity loss reduces the efficiency of ecological communities; (ii) biodiversity makes more stable ecosystem functions over the time; (iii) the impact of external changes increases as biodiversity decreases; and (iv) differences among organisms (i.e. functional traits) increase the whole ecosystem stability and efficiency. Biodiversity fits the concept of ecosystem services in at least two ways, as follows: (i) biodiversity and ecosystem services are the same thing (ecosystem services perspective); and (ii) biodiversity has an existence value (conservation perspective). For example, Gamfeldt *et al.* (2013) demonstrated that forests with more tree species have a positive relationship with the delivering of multiple services, as well as highlighted that conserving a variation of species is fundamental to safeguard a future potential of high levels of ecosystem services provision (i.e. no single species can sustain multiple services at high levels simultaneously). There is still poor understanding on how biodiversity effectively supports the provision of other forest ecosystem services (see §2.3). Table 30 summarizes the linkages between resilience, biodiversity and some forest ecosystem services.

Table 30: Summary of the most important linkages between resilience, biodiversity and forest ecosystem services (Hicks *et al.* 2014, modified).

Forest ecosystem service group	Forest ecosystem service type	Highlights	Selected references
Regulating services	Carbon sequestration and storage (i.e. climate change mitigation)	• Biodiversity, intactness and naturalness affect forest carbon stock resilience • Different components of biodiversity, including identity, relative abundance, number and spatial arrangement of species in principle probably have an impact on stability and predictability of carbon stocks • Carbon stocks of intact forests are more resilient than those of degraded or fragmented forests. More varied species composition in natural forests appears to increase regeneration compared to plantation forest	Miles *et al.* (2010); Conti and Díaz (2013); Bunker (2005); Lawrence *et al.* (2005)
	Soil erosion control	• Intact forest cover prevent rapid runoff, thus reducing the susceptibility of the land to extreme erosion phenomena • Vegetation structure and plant life forms are the main factors responsible for reducing surface runoff and the movement of sediments	Watkins and Imbumi (2007); Zhao *et al.* (2009)

Forest ecosystem service group	Forest ecosystem service type	Highlights	Selected references
	Soil fertility and nutrients	• Species richness is important on the basis that the leaf litter of different tree species plays different roles in improving soil fertility, depending on their "quality" or chemical compositions • Vegetation presence, biomass and types (rather than for species richness or other aspects of biodiversity) have benefits for preventing soil erosion or nutrient loss • total phosphorus loss decreases with increasing plant species richness	Vityakon (2001); Wang *et al.* (2007)
	Pollination	• Forest loss causes negative impacts on potential pollinator communities and seed sets of some woodland plants • The presence of large forest patches in diversified landscapes is associated with abundance of bees	Taki *et al.* (2007); Brosi *et al.* (2008)
	Water quantity and quality	• Shifts in the hydrological regime are associated with human-induced changes in vegetation type and density, most likely to be related to the conversion of native forests to agricultural land	Molina *et al.* (2012)
	Protection from natural hazards	• Trees make a significant mechanical contribution to reducing shallow landslide development during a severe storm event in steep, forested watersheds	Kim *et al.* (2013)
Provisioning services	Timber production	• Timber from areas with high numbers of species with timber utility are used for a greater variety of purposes • Timber extraction is higher in forests with lower biodiversity and *vice versa* • Reducing the impact of logging on forest biodiversity may improve the long-term productivity of the forest through improving regenerative capacity of the forest, and reducing vulnerability to fires (through reduced organic debris)	Njana *et al.* (2013); Chopra and Kumar (2004)
	Non-timber forest products (NTFPs)	• High functional redundancy (several species can be used for the same purpose) is an important factor defining the value of a forest as a source of NTFPs; in a forest with high functional redundancy, changes in species richness do not immediately lead to the loss of use value	Brown *et al.* (2011)
Tourism and cultural services	Recreational, spiritual, aesthetic, educational, etc.	• Biodiversity plays an important role in fostering a sense of place in most communities living or visiting diversified forested landscapes	Fuller *et al.* (2007); Price *et al.* (2011); Naughton-Treves

Forest ecosystem service group	Forest ecosystem service type	Highlights	Selected references
		• Biodiversity in urban forest areas plays a positive role in enhancing human well-being and providing psychological benefits • Tree species richness, presence of habitat diversification, and conservation of wildlife in general (i.e. as in Protected Areas) increase the number of visits for tourism or recreational purposes	*et al.* (2005)

5.1.1 Management strategies to improve resilience in forest stands

Managing forests to improve the resilience of BES requires a deeper understanding of ecosystem functioning and of the interactions among species and habitat types at different scales. Relationships between biodiversity (in terms of e.g. species richness, structural diversity, etc.) and other forest ecosystem services need to be further explored and monitored in order to improve forest resilience over the time (see e.g. Gardner 2010). For example, results from case study 3 (see §3.4) demonstrate that stand age distribution, turnover rate, regeneration capacity, and natural mortality rates are some of the most important stand characteristics influencing the biodiversity conservation among future-oriented scenarios. Accordingly, Mace *et al.* (2012) suggested that: (i) biodiversity (as a regulator of ecosystem processes) needs to be managed in order to achieve productivity and maintain nutrient cycles and decomposition but with the high risk of reducing the ability of the system to deliver other services (as with regards to timber production; see §3.4); and (ii) biodiversity (as a final ecosystem service) needs to be managed to maintain the necessary range of species groups and habitat or landscape types, with profound implications for management practices, especially if the maximizing of one service is a management goal. At stand level, reducing harvesting rates (i.e. intensity and frequency of intervention), prolonging rotation periods, and increasing the amount of timber releases are important forest management strategies for maintaining forest biodiversity and ecosystem resilience at higher levels in the future (see §3.4; for resilience in carbon stock, see §4.3). These findings are consistent with the ecological forestry guide-lines (Franklin *et al.* 2007), which are summarized in Table 31.

Table 31: Main ecological principles and related management guide-lines to improve resilience of forest ecosystems (Franklin *et al.* 2007, modified).

Ecological principles	Management guide-lines
Incorporating biological legacies[9] into harvest prescriptions	Incorporating spatial heterogeneity of retention within a harvest unit, by modifying clearcut and shelterwood prescriptions to include wildlife trees, snags, and logs (mainly Two-Cohort and Selection Systems). In marking structures for retention, attention should be given to retaining a diversity of tree species. Emphasis should be given to retaining trees across a range of size classes and levels of decadence. In fact, retaining

[9] Biological legacies are defined as the organisms, organic matter (including structures), and biologically created patterns that persist from the pre-disturbance ecosystem and influence recovery processes in the post-disturbance ecosystem (Franklin and MacMahon 2000). Legacies occur in varied forms and densities, depending upon the nature of both the disturbance and the forest ecosystem (Franklin *et al.* 2007).

Ecological principles	Management guide-lines
	trees in various states of decadence is important for providing critical habitat features, such as cavities, and to ensure a sustained source of large dead wood (see e.g. Burrascano *et al.* 2008).
Intermediate treatments that enhance heterogeneity	The primary way is to adopt innovative thinning approaches in order to: (i) simulate the development of larger trees; (ii) simulate the development of horizontal heterogeneity (Variable Density Thinning; VDT, Carey 2001); and (iii) develop vertical and horizontal heterogeneity (small gap creation). In the first case, Appropriately implemented thinning from below accelerates the development of large-diameter and high-quality trees at rates faster than would occur naturally. In the second case, VDT generates much greater spatial variability in stand densities and, consequently, greater structural complexity and heterogeneity of structure. In the third case, the creation of small gaps in forest canopy generates opportunities for establishing and releasing regeneration and other understory components.
Allowing for appropriate recovery periods	Allowing for appropriate recovery periods between management entries, especially regeneration harvests (in which case the recovery period is traditionally known as the rotation), to allow complexity to develop. Although recovery periods are almost always much longer than rotations based on economic factors and probably longer than rotations based on growth factors, culmination of growth increment can be delayed for extended periods of time, through periodic thinning.

Forest management approaches going towards the resilience thinking move along a complexity continuum with different levels of randomness (from order to chaos) (see Figure 38).

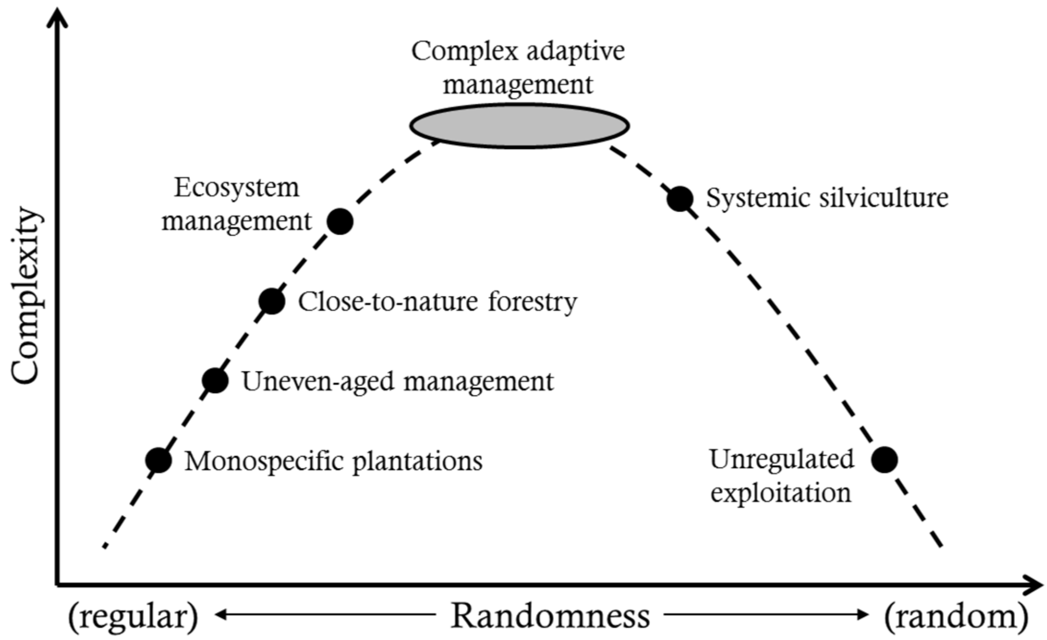

Figure 38: Convex relationship between complexity and regularity of patterns. Different types of forestry approaches are aligned along the gradient of regularity. The diagram suggests in which direction along the regularity gradient these management approaches would have to move to increase complexity in managed forests (Parrott 2010, modified).

Managing forests in or near the middle zone (between order and chaos; Figure 38) is conceptually desirable because complex systems are thought to be more resilient, better able to adapt to rapidly changing conditions and more likely to provide the numerous and varied services that people desire and need to prosper and continue on this planet (Loreau *et al.* 2001; Carlson and Doyle 2002; Hooper *et al.* 2005; Levin 2005). Moreover, managing forests to improve resilience has to be based on heterogeneity, diversity and variability characteristics of forest ecosystems (Messier *et al.* 2013). Further efforts are necessary to promote self-organization and adaptive capacity of forest ecosystems through practical forest management (Ciancio and Nocentini 2011), while recognizing that the outcome of any management practice is inherently high uncertain (see e.g. Lindner *et al.* 2014).

5.1.2 Management approaches to improve resilience in forest landscapes

Complex adaptive systems typically contain feedbacks and non-linearities, as well as possessing the capacity to self-organize; manipulations can have surprising and unintended consequences (e.g. Foley *et al.* 2003). For these reasons, successful manipulations of complex adaptive systems have been built on bottom-up approaches for adaptation and learning rather than imposing a particular planning or management goal from the top-down (e.g. Bohensky 2008). Landscape resilience depends heavily on finding an appropriate match between the scales of demands on ecosystems by human societies and the scales at which ecosystems are capable of meeting these demands (Cumming *et al.* 2006). The most effective way to move towards sustainable landscapes appears to be to deliberately encourage local and regional social-ecological experiments that allow social learning to occur within the context of finding long-term solutions to chronic, broad-scale problems (Cumming *et al.* 2013). Both long-term monitoring and the creation and implementation of diversity in problem-solving approaches rely on adaptive governance and management approaches that: (i) stimulate social learning by involving actors at multiple levels, from local to global; (ii) support the translation and diffusion of new knowledge and practices, creating a continuous feedback between research and implementation and potentially transforming societal attitudes and motivations (cf. §4.2); and (iii) offer ''safety nets'' to communities that are willing to engage in potentially risky experimentation (Cumming *et al.* 2013).

Considering that ecosystem resilience partly follows the conservation biology principles (ecosystem integrity, structural complexity, and habitat connectivity; Voller and Harrison 2011), forest management of European forest landscapes is oriented to (Bollmann and Braunisch 2013): (i) preserve rare, representative, and threatened forest types or stands, such as the last remaining pristine and ancient forests, as well as the retention of old or old-growth stands, mature trees, and coarse woody debris (CWD) within managed forest landscapes; (ii) restore important habitats and structural characteristics by constitutive measures (e.g. creating gaps, controlled burning and browsing, ring barking, uprooting of trees); and (iii) support natural (succession) dynamics after disturbance events. Managing resilient forest landscapes can be realized by adopting a segregative or integrative approach. A strictly segregative approach allocates a certain ratio of the landscape for nature conservation (e.g. forest reserve), while commodity production is maximized in the remaining landscape. In contrast, a strictly integrative approach aims at combining ecological, economic, and social issues across the total forest area at the same time. Table 32 reports the differences between the most important segregative and integrative instruments.

Table 32: Definition of the integrative and segregative instruments (Bollmann and Braunisch 2013).

Conservation instrument	Purpose	Forest Management Approach
National park	Designated landscape area according to IUCN protected area management categories in order to preserve unique ecosystems with native species and communities under natural dynamics to enable their long-term viability.	Segregative
Strict forest reserve	Protected forest area aiming for biodiversity conservation by natural dynamics with no or minimal human intervention.	Segregative
Special forest reserve	Protected area aiming at enhancing forest biodiversity through active habitat restoration or management, such as prescribed burning, cutting and mowing, controlled grazing and browsing, and rebuilding of coppice with standards.	Segregative
Biosphere reserve	Established areas designated under UNESCO's Man and the Biosphere (MAB) Programme to promote sustainable development by a zonal concept based on local community efforts and evidence-based conservation.	Segregative, (Integrative)
Structural retention	Retention of key structural habitat elements such as habitat trees, snags, lying deadwood, gaps, and riparian stands in commercially used forests.	Integrative
Old-growth stand protection	Protection of old-growth stands with mature and dead trees as habitat patches and stepping stones in commercially used forests.	Integrative
Wildlife corridor	Site traditionally used by wildlife species to move between populations separated by human activities or structures such as highways, urban development, and clearcuts.	Integrative
Ecological process area	Temporally restricted and spatially flexible conservation instrument that integrates natural dynamics and its habitat features after a disturbance event in production forests for some decades. Later, the area is re-integrated and managed again according to the purposes of regional forestry until a consecutive disturbance occurs.	Integrative

In recent years growing evidence has emerged that large-scale forest biodiversity conservation depends on a combination of both approaches (Bengtsson *et al.* 2003), especially since the impact of the various tools and the responses to their application are scale-dependent. For example, evidences from case study 4 (see §4.2) demonstrate that in Italy the network of National and Regional Parks plays an active role in conserving forest biodiversity and preserving the delivery of all forest ecosystem services (see also Schirpke *et al.* 2014; Marchetti *et al.* 2012). Especially in Protected Areas, biodiversity conservation and habitat integrity are important drivers for improving the wellbeing of local communities and enhancing tourism and recreational opportunities. A concept with a dual strategy combining integrative and segregative instruments seems to be the best option to support biodiversity conservation in a cultural landscape, with a system of multi-purpose forestry and variation in forest tenure (see Figure 39; Bollmann and Braunisch 2013). An important field of research remains how the combination of complementary instruments in a qualitative and spatially optimized way may support ecosystem functions that cannot be supported with one type of instrument alone.

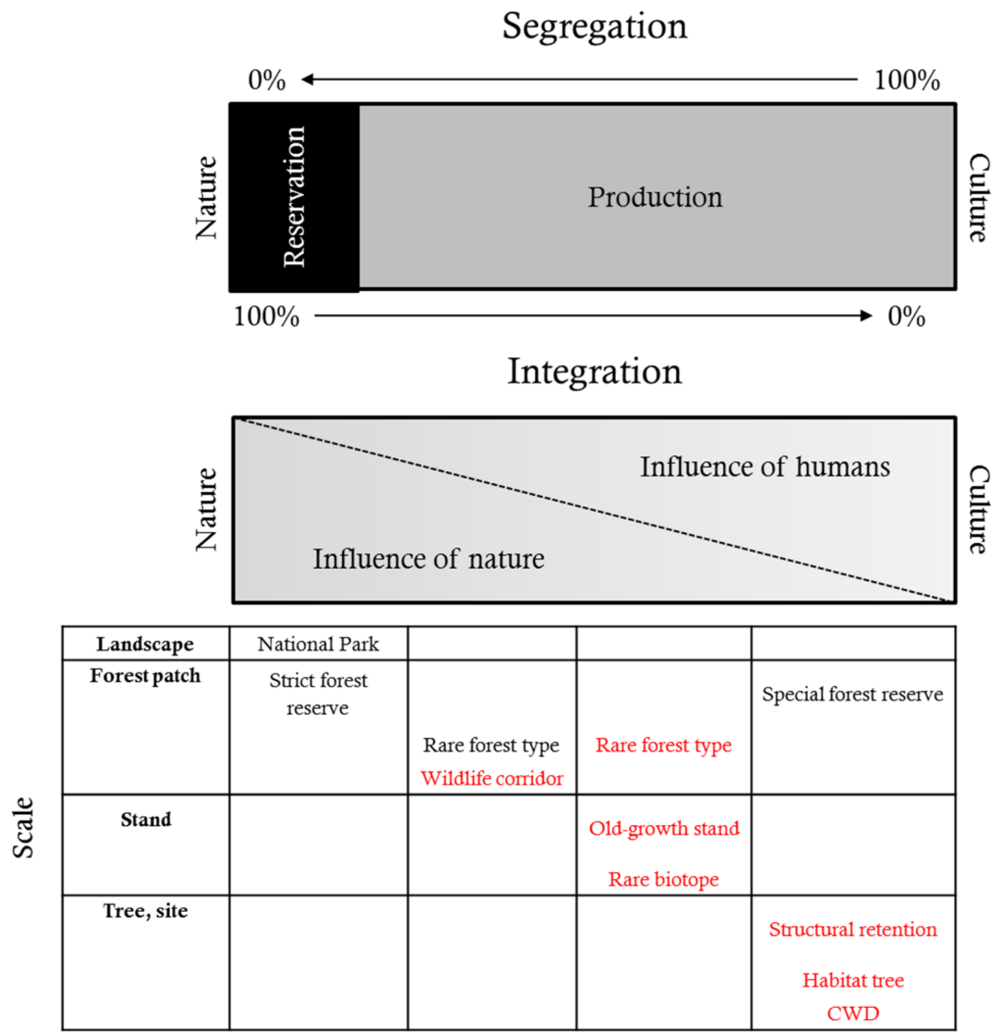

Figure 39: Conceptual differences between segregative and integrative approaches in forestry (Bollmann and Braunisch 2013, modified). In a segregative forestry system, national parks and forest reserves often preserve primeval or heritage forests that are embedded in a matrix of intensively used forests or plantations with low habitat quality. In a purely integrative system, structural retention and restoration measures (red) are an integral part of sustainable forest practices. They mainly support minimum targets of habitat features and resources, but their impact is mostly restricted to the site and stand scale. In an optimized integrative system, these small-scale conservation measures are combined with segregative tools (black). They often support ecological process dynamics at the forest patch and landscape scale as targeted by national parks or strict forest reserves. Yet, segregative tools can also be used to actively restore traditional forest habitats for specific conservation purposes (e.g. special forest reserve). Integrative forestry systems such as those in Central Europe often lack remnants of primeval forest at the very left side (darker grey) of the nature-culture gradient (see Winter *et al.* 2010).

At a broader scale, ecosystem services assessment makes conservation plans more effective through the following ways (Egoh *et al.* 2007): (i) Payments for Ecosystem Services (PES) are potentially a strong avenue for securing priority areas (Engel *et al.* 2008); (ii) services have an advantage in that they are linked to beneficiaries and thus facilitate the implementation of conservation plans; and (iii) targeting services in conservation assessments may achieve many biodiversity targets under an easy-to-sell umbrella of ecosystem services while at the same time improving the relevance of conservation plans to human wellbeing. Nevertheless, some constraints have to be considered while planning for BES, such as (Egoh *et al.* 2007): (i) the discordance between priority biodiversity features and spatial features required for ecosystem services delivery; and (ii) the different values related to biodiversity and ecosystem services (i.e. intrinsic vs. utilitarian, respectively), which require different stakeholders and agencies (i.e. conservation agencies vs. resource managers, respectively).

Finally, the cooperation between scientists and policy makers should be promoted. In particular, much more efforts are required to translate science outcomes into policy strategies, such as (Thompson *et al.* 2011): (i) the clarification of the mechanisms by which biodiversity supports and maintains ecosystem goods and services and the clear illustration of these mechanistic effects, (b) improvement of the valuation methods of these ecosystem services to human society, and (c) the derivation of meaningful values (target ranges) and known thresholds to improve the usefulness of biodiversity indicator. Results from case study 4 (see §4.2) demonstrate that there is still weak cooperation and scarce exchange of information between management authorities, researchers, and local stakeholders, at least in Protected Areas. Therefore, conservation biologists and ecosystem managers need to work together to effectively implement resilience thinking objectives (Mace *et al.* 2012).

5.3 Understanding the key-role of community-based management to improve forest ecosystem resilience

Complex adaptive systems originate from the interactions between people and ecosystems (e.g. Liu *et al.* 2007). In this sense, Community-Based Natural Resources Management (CBNRM) plays a key-role in enhancing ecosystem resilience because (Davidson-Hunt and Berkes 2003): (i) management practices are locally adapted and based on local ecological knowledge (e.g. 'Traditional Forest-Related Knowledge'; Trosper and Parrotta 2012); (ii) local institutions are "close to the ground" and able to observe and adapt rapidly, making and learning from small mistakes where centralized bureaucracies make large ones (Agrawal 2007); (iii) there is a tremendous diversity among local CBNRM groups, and such diversity increases the 'learning-by-doing' approach (Berkes 2009); (iv) CBNRM is able to strengthen social capital, which is a key driver to promote adaptive capacity of local communities (Adger 2003; Walker and Salt 2006; Armitage 2005); and (v) CBNRM promotes social learning, an intentional process of collective self-reflections through interaction and dialogue among diverse participants (Keen and Mahanty 2006).

At local scale, communities are proximal to the resources they use. As a consequence, the whole community or selected individuals (e.g. stewards or elders) can monitor the status and observe day-to-day changes of ecosystems (Berkes *et al.* 2000). On the other hand, using knowledge and perspectives from the community level can help build a more complete information base that may be available from scientific studies alone (Berkes *et al.* 2000). It is also demonstrated that traditional management systems contribute to the conservation of biodiversity, through diversifying the use of more varieties, species (mostly native), and landscape patches than do modern agricultural and food production systems (e.g. Berkes 2004). Biodiversity evolved in the context of human use and depends on it. It is evidenced by the fact that the world's most biodiverse regions are also the world's most culturally diverse regions (e.g. Anderson 2005). On historical basis, the traditional use of resources ensures the maximization of ecosystem services provision, while maintaining ecosystem stability and health at high sustainability levels (e.g. Berkes 1989). Table 33 summarizes the traditional socio-ecological practices to improve ecosystem resilience.

Table 33: Traditional social and ecological practices and mechanisms to improve resilience and sustainability (Folke *et al.* 1998, modified).

Management practice groups	Management practice types	Implementation
Management practices based on ecological knowledge	Practices found both in conventional resource management and in some local and traditional societies	• Monitoring resource abundance and change in ecosystems • Total protection of certain species • Protection of vulnerable life history stages • Protection of specific habitats • Temporal restrictions of harvest
	Practices largely abandoned by conventional	• Multiple species management;

Management practice groups	Management practice types	Implementation
	resource management but still found in some local and traditional societies	maintaining ecosystem structure and function • Resource rotation • Succession management
	Practices related to the dynamics of complex systems, seldom found in conventional resource management but found in some traditional societies	• Management of landscape patchiness • Watershed-based management • Managing ecological processes at multiple scales • Responding to and managing pulses and surprises • Nurturing sources of ecosystem renewal
Social mechanisms behind management practices	Generation, accumulation, and transmission of local ecological knowledge	• Reinterpreting signals for learning • Revival of local knowledge • Folklore and knowledge carriers • Integration of knowledge • Inter-generational transmission of knowledge • Geographical diffusion of knowledge
	Structure and dynamics of institutions	• Roles of stewards/wise people • Cross-scale institutions • Community assessments • Taboos and regulations • Social and religious sanctions
	Mechanisms for cultural internalization	• Rituals, ceremonies, and other traditions • Cultural frameworks for resource management
	World view and cultural values	• A world view that provides appropriate environmental ethics • Cultural values of respect, sharing, reciprocity, humility, and other

The southern region of Europe has seen the abandonment of many traditional forestry practices, such as coppicing for firewood and the collection of barks, resins, acorns and tannin, often as a consequence of rural depopulation (for the Italian peninsula, see Agnoletti 2007). Community involvement in forest protection in Europe takes many forms, from financial support to woodland conservation organizations and charities; local initiatives concerned with native woodland management and conservation, and direct actions (Jeanrenaud 2001). In Italy, public participation experiences in decision-making processes are still very scarce (Cantiani 2006; see also §4.2).

Local communities have traditional rights to use public forests for collection of wood and other products (Marinelli 2013). There are specific regional rules for hunting, and harvesting NWFPs such as mushrooms, truffles, pine seeds, chestnuts and cork – which make significant contributions to local economies. Traditional rights are usually promoted by representatives in local and national governments owning public forests. It is hard to eliminate such uses even for conservation purposes, so they are usually maintained even in National Parks (Jeanrenaud 2001). So far, community forestry has been an important element enhancing ecosystem resilience and sustainable development (Charnley and Poe 2007), especially in many Northern Italian regions, mainly due to the combination between political autonomy, strong social ties and community welfare. Over the last few centuries, political, economic and social changes have strongly limited the institution of community forestry in many regions in Italy. This was not the case of stronger regions (in terms of available financial resources), where community forestry contrasted the trends towards greater state control and privatization of forests (e.g. "Val di Fiemme"; Morandini 1996). This aspect demonstrates that community forestry has a high degree of dynamism and flexibility in the face of social and economic change.

To improve ecosystem resilience, co-management[10] (Carlsson and Berkes 2005) is considered a suitable strategy. Co-management focuses on several aspects characterizing complex adaptive systems, such as issues of scale, multiple perspectives and epistemologies, path dependence, and uncertainty (Berkes 2007). Effective co-management requires flexible, multi-level governance systems designed to enhance institutional interaction and experimentation to generate learning (Folke *et al.* 2002; Kooiman *et al.* 2005), but there is little experience on how to accomplish this (Berkes 2009). Table 34 lists some important strategies that have been used to facilitate or improve co-management, and that can be further use to improve ecosystem resilience through community forestry.

Table 34: Strategies that have been used to facilitate or improve co-management (Berkes 2009, modified).

Strategies	Description	Selected references
Bridging knowledge	Incorporating multiple knowledge systems and multiple scales enhances environmental decision-making	Eamer (2006)
Co-production of knowledge	Researchers/scientists working with place-based learning communities can co-produce locally relevant knowledge that neither party can produce alone	Davidson-Hunt and O'Flaherty (2007)
Participatory research	Research that includes rural and indigenous communities as equal partners has the potential to build social capital and enhance local capacity for problem solving	Arnold and Fernandez-Gimenez (2007)
Collaborative monitoring	Monitoring that includes, where possible, local ways of reading environmental signs and signals have the potential to widen the range of information available	Kofinas (2002)
Participatory	Scenario building that includes joint deliberation about what is known	Bennett and Zurek

[10] Co-management (i.e., cooperative management) is based on broad levels of cooperation. It relies on ''the collaboration of a diverse set of stakeholders operating at different levels, often in networks, from local users, to municipalities, to regional and national organizations'' (Olsson et al. 2004). An integrating term, ''adaptive co-management'', combines the dynamic learning characteristics of adaptive management with the collaborative networks inherent in co-management.

Strategies	Description	Selected references
scenario building	and what is not known provides an ideal space about questioning assumptions made by different disciplines and different perspectives	(2006)

5.4 Towards the "resilience thinking": key messages

Complex adaptive systems (e.g. Mediterranean forests) are increasingly threatened or degraded by human-induced disturbances (such as e.g. climate change effects) and need to be managed in a way (and at a rate) that is suitable to promote their resilience, resistance, and stability in the future. In turn, the decreasing of resilience in forest ecosystems affects the conservation of biodiversity and the delivery of fundamental benefits for local communities.

Forest management has an impact on forest ecosystem resilience, through increasing or reducing the capacity of forests to face external changes. Moreover, forest management practices (in terms of frequency and intensity of interventions) directly influence biodiversity conservation and services provision over the time, by modifying e.g. ecosystem structures, biological legacies, landscape assets, etc.

Understanding forest ecosystem functioning, as well as the linkages between biodiversity conservation and the delivery of additional goods and services is fundamental to improve forest resilience through adaptive management. In this way, monitoring, assessing, and mapping forest ecosystems and their services may be useful to define future-oriented management guide-lines by a holistic perspective. In addition, forest simulation tools and modeling techniques must be further developed through implementing robust indicators and consistent parameters describing external changes and disturbances, such as e.g. climate parameters, people perceptions, land use transitions, other landscape and environmental barriers/drivers etc.

Implementing the "resilience thinking" in forest management requires a strong cooperation between policy-makers, managers, stakeholders, and local communities. At first, research findings have to be exchanged with local managers and stakeholders, in order to stimulate public interest in trans-disciplinary issues (i.e. resilience-based theory and purposes). Secondly, participation of local communities in decision-making processes must be encouraged and promoted at different scales. Finally, traditional ecological knowledge of local communities can be used as a basis for adaptive forest management and for improving ecosystem resilience at a local scale.

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List of Figures

Figure 1: Unified framework for integrating Millennium Development Goals (MDGs) with sustainability conditions in the context of revised SD definition. Source: Griggs et al. 2013, modified. 3

Figure 2: Evolutionary flowchart reporting the key definitions of Integrated Forest Management, Ecosystem-based Management, and Adaptive Forest Management as means to maintain sustainability of forest management. 8

Figure 3: Conceptual description of planetary boundaries. The equilibrium response is plotted as a function of the strength of multiple, interacting anthropogenic drivers, such as overharvesting or ocean acidification. Uncertainty over the eventual impact of high levels of drivers is indicated by considering two potential system responses at equilibrium: smooth, and hysteretic (or folded). The latter constitutes threshold effects (Hughes *et al.* 2013, modified). 11

Figure 4: The FES framework (Vizzarri *et al.* 2013). The chart summarizes the most important FES (such as supporting, provisioning, regulating and cultural). Red and blue arrows explain, respectively, the *in-out* relationships (or effects) occurring between forest ecosystems and other systems (anthropogenic and other natural and semi-natural) following a holistic approach. The large arrows on background describe the direction of energy and material fluxes from supporting services (viewed here as ecosystem functions or intermediate services), which move throughout the forest biophysical structures and process, and finally originate the other final services. Adversely, semi-circular arrows represent the benefits originated by all final services on both anthropogenic and other natural and semi-natural systems. The FES model is bordered in blue. It details the relations between natural and human contexts. In particular, the FES model highlights that, while FES framework describes the fluxes of tangible and non-tangible goods and products from forests to people (as benefits), the sustainable forest management (SFM) ensures and enhances the exploitation of such benefits, as well as it describes human interventions and the anthropogenic effects on forest ecosystems. Moreover, as showed in the FES model, both FES and SFM can be assessed and measured though adopting and implementing several indicators. 31

Figure 5: From left to right, climate service in (A) tropical, (B) temperate, and (B) boreal forests. Text boxes indicate key processes with uncertain climate services. Source: Bonan (2008a), modified. 31

Figure 6: Summary of the effects of vegetation in minimizing the erosion (Menashe *et al.* 1993, modified). 33

Figure 7: The 'service cascade' representation. Sources: Haines-Young and Potschin (2010); de Groot *et al.* (2010a); Kandziora *et al.* (2013), modified. 38

Figure 8: Approaches for the estimation of nature's values. Source: Gómez-Baggethun and de Groot (2010), modified. 40

Figure 9: Trends of the n pub year^{-1} as resulted in RS A and for SS A.1 (a), SS A.2 (b), and SS A.3 (c). 45

Figure 10: Trends of the n pub year^{-1} as resulted in RS B and C and for: (a) SS B.1 and C.1; (b) SS B.2 and C.2; (c) SS B.3 and C.3; and (d) SS B.4 and C.4. 46

Figure 11: Number of publications per TA as resulted in RS D. 48

Figure 12: DSS architecture (Eriksson and Borges 2014, modified). 67

Figure 13: FLMP pilot studies and regional implementation distributions in Italy (a), and a zoom-on both Molise region and case study area (b). Numbering of FLMP pilot studies as follows: (1) the Cadore, Longaronese, and Zoldo MC; (2) the Asiago plateau MC; (3) the Piné plateau; (4) the watershed of Trasimeno lake; (5) the Trigno-Biferno rivers MC; (6) the Alto Molise MC; (7) the Agri river plain area and the "Appennino Lucano-Val d'Agri-

Lagonegrese” National Park; (8) the Collina Materana MC; (9) the Natural Reserve of Sosio valley and Palazzo Adriano mountains; (10) the North-western area of Etna mountain; and (11) the “Arci-Grighine” district. References in the text. 81

Figure 14: Sampling plots distribution within the study area. 83

Figure 15: The MCML approach. 87

Figure 16: Box-plot showing the distribution of the collected forest attributes between the assigned FDUs. 88

Figure 17: G, TSN and H_r maps as resulted by the *k*-NN spatialisation process (a, b, and c, respectively). 89

Figure 18: FDUs map. 90

Figure 19: FES trade-offs between scenarios per each case study at the end of simulation period (30 years from now). FES values are reported in terms of the ratio between the FES value of a given scenario and the maximum FES value for all considered scenarios (ES/max ES). 100

Figure 20: Linear regressions between forest ecosystem services and impact factors for the three case studies (from top to bottom, ASI, COM, and ETN) at the end of simulation period (30 years from now). 103

Figure 21: The restoration staircase. Depending on the state of degradation of an initially forested ecosystem, a range of management approaches can at least partially restore levels of biodiversity and ecosystem services given adequate time (years) and financial investment (capital, infrastructure, and labor) (Source: Chazdon 2008). 122

Figure 22: Bar chart reporting the FES relevance (grouped by Division) in surveyed NRPs. Values vary from very low (0) to very high (4) relevance. Errors bars refer to SD values. 128

Figure 23: Spider charts showing the different impacts of stakeholder typologies on FES, lettered from A to H as follows: NWFPs/fresh water availability (A); Production of raw materials (B); Timber extraction for energy supply (C); Bioremediation (D); Hydrogeological protection (E); Biodiversity conservation and climate change mitigation (F); Improvement of tourism and recreation concerns (G); and Conservation of the landscape identity (H) (see Appendix 2 for the nomenclature). Values range from -1 (negative impact) to 1 (positive impact). Dashed lines represent the 0 value. 129

Figure 24: Percentage of NRPs where governance instruments are currently at work. 130

Figure 25: Bar chart showing the relative percentage of respondents to several sentences regarding the role of different factors (local stakeholders, FES analysis, etc.) in decision-making processes. 131

Figure 26: Bar chart showing the relative percentage of respondents to several sentences regarding the role of research (related activities, results and advances) in decision-making processes. 132

Figure 27: Bar chart reporting the ranking values in assessing FES priority, difficulty and relevance in forest management. Error bars represent the Standard Deviation (SD) values. 133

Figure 28: Flowchart reporting carbon fluxes/processes (arrows) and stocks (grey boxes) in a forest ecosystem, and as simulated by CO2FIX model (Schelhaas et al. 2004). 143

Figure 29: 300-year simulation of C stock in above-ground biomass for Mediterranean and Anatolian black pine forest managed with clearcutting system. 145

Figure 30: 300-year simulation of C stock in above-ground biomass for Mediterranean-Anatolian silver fir even-aged forest managed with clearcutting system. 146

Figure 31: 300-year simulation of C stock in above-ground biomass for montane beech forest managed with shelterwood system. 147

Figure 32: 300-year simulation of C stock in above-ground biomass for montane beech, high-coppice forest, managed by conversion of ageing coppice stand and then with shelterwood system. 148

Figure 33: 300-year simulation of C stock in above-ground biomass for Mediterranean evergreen oak, high-coppice forest managed by conversion of ageing coppice stand and then with shelterwood system. 149

Figure 34: 300-year simulation of C stock in above-ground biomass for Subalpine and mountainous spruce-silver fir mixed, uneven-aged forest managed with selection system. 150

Figure 35: 300-year simulation of C stock in above-ground biomass for Thermophilous pine, even-aged forest managed with clearcutting system. 151

Figure 36: 300-year simulation of C stock in above-ground biomass for Montane beech, high-coppice forest in CAP managed by conversion of ageing coppice stand and then with selection system. 152

Figure 37: 300-year simulation of C stock in above-ground biomass for Montane beech, uneven-aged forest in ASI managed with selection system. 153

Figure 38: Convex relationship between complexity and regularity of patterns. Different types of forestry approaches are aligned along the gradient of regularity. The diagram suggests in which direction along the regularity gradient these management approaches would have to move to increase complexity in managed forests (Parrott 2010, modified). 173

Figure 39: Conceptual differences between segregative and integrative approaches in forestry (Source: Bollmann and Braunisch 2013, modified. In a segregative forestry system, national parks and forest reserves often preserve primeval or heritage forests that are embedded in a matrix of intensively used forests or plantations with low habitat quality. In a purely integrative system, structural retention and restoration measures (red) are an integral part of sustainable forest practices. They mainly support minimum targets of habitat features and resources, but their impact is mostly restricted to the site and stand scale. In an optimized integrative system, these small-scale conservation measures are combined with segregative tools (black). They often support ecological process dynamics at the forest patch and landscape scale as targeted by national parks or strict forest reserves. Yet, segregative tools can also be used to actively restore traditional forest habitats for specific conservation purposes (e.g. special forest reserve). Integrative forestry systems such as those in Central Europe often lack remnants of primeval forest at the very left side (darker grey) of the nature-culture gradient (see Winter *et al.* 2010). 176

Supplementary material

Appendix 1

Review section E: list of EU funded projects

Table A1.1: List of EU-funded projects (and related details) about forest ecosystem services for the 2000-2012 reference period.

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
ALARM	Assessing LArge-scale environmental Risks with tested Methods	01/02/2004	31/01/2009	http://www.alarmproject.net/alarm/	Yes	YES	NO	Public
ALTER-NET	A Long-term Biodiversity, Ecosystem and Awareness Research Network	01/04/2004	31/03/2009	http://www.alter-net.info/	Partially	YES	NO	Public
AMECO	Assisted Migration of Forests as a climate change economic mitigation strategy	01/05/2013	30/04/2015	NF	NA	NO	NO	Public
ARANGE	Advanced multifunctional forest management in European	01/02/2012	31/07/2015	http://www.arange-project.eu/	Yes	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	mountain ranges							
ASEC-DRYLAND-FORESTS	Avoiding the socio-ecological collapse of remnant evergreen forests in drylands: the case study of northern Kenya	NA	NA	NF	NA	NO	NO	Public
ATEAM	Advanced terrestrial ecosystem analysis and modelling (ATEAM)	01/01/2001	30/06/2004	https://www.pik-potsdam.de/ateam/	Yes	NO	NO	Public
CARBOAFRICA	Quantification, understanding and prediction of carbon cycle, and other GHG gases, in Sub-Saharan Africa	01/10/2006	30/09/2009	http://www.carboafrica.net/index_en.asp	Not specifically	YES	YES	Public
CCTAME	Climate change - terrestrial adaption and mitigation in Europe	01/06/2008	31/08/2011	http://www.cctame.eu/	Partially	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
CLIMSAVE	Climate change integrated assessment methodology for cross-sectoral adaptation and vulnerability in Europe	01/01/2010	31/10/2013	http://www.climsave.eu/climsave/index.html	Partially	NO	NO	Public
COMDREEF	Community disassembly rules and the erosion of ecosystem functions in fragmented landscapes	NA	NA	NF	NA	NO	NO	Public
CRUE	Coordination de la Recherche sur la gestion des inondations financie dans l'Union Europene (Coordination of research financed in the European Union on Flood risk	01/11/2004	31/10/2009	NF	NA	YES	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	management)'							
DESURVEY	A Surveillance System for Assessing and Monitoring of Desertification	11/03/2005	10/10/2010	http://www.noveltis.com/desurvey/	Not specifically	YES	NO	Public
DYVERSE	Vegetation dynamics and ecosystem services provision in a fragmented landscape in response to global change	01/06/2011	31/05/2014	NF	NA	NO	NO	Public
ECOADAPT	Ecosystem-based strategies and innovations in water governance networks for adaptation to climate change in Latin American Landscapes	15/01/2012	14/01/2016	https://sites.google.com/site/ecoadaptprojectenglish/	Not specifically	NO	NO	Public
ECOOP	European COastal-shelf sea OPerational	01/02/2007	30/04/2010	http://www.ecoop.eu/	Not specifically	YES	NO	both

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	observing and forecasting system							
EFORWOOD	Tools for Sustainabiltity Impact Assessment of the Forestry-Wood Chain	01/11/2005	31/01/2010	http://www.innovawood.com/eforwood/	Partially	NO	NO	Public
EU-MEDIN COMPANIONS	Supporting publications on Natural Hazards Research	01/07/2005	30/11/2007	NF	NA	NO	NO	Public
FIRE PARADOX	An innovative approach of Integrated Wildland Fire Management regulating the wildfire problem by the wise use of fire: solving the FIRE PARADOX	01/03/2006	28/02/2010	http://www.fireparadox.org/index.php	Not specifically	YES	NO	Public
FIXSOIL	Understanding how plant root traits and soil microbial	01/05/2014	30/04/2016	NF	NA	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	processes influence soil erodibility							
FLAGSHIP	European Framework for safe, efficient and environmentally - friendly ship operations	01/01/2007	31/05/2011	http://flagship-project.eu/	Not specifically	YES	NO	private
FLUORFLIGHT	FluorFLIGHT: A new integrated canopy fluorescence model based for remote sensing of forest health and productivity	NA	NA	NF	NA	NO	NO	Public
FORADAPT	Decision support toolkit FOR ADAPTive management of forest ecosystem services across borders in the face of climate change and	01/02/2015	31/01/2017	NF	NA	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	economic scarcity in Europe							
FORCONEPAL	Forest Resource Conservation in Nepal	NA	NA	NF	NA	NO	NO	Public
FORECOFUN-SSA	Assessing climate change impacts over large areas of primary forests in southern South America	01/03/2012	28/02/2014	NF	NA	NO	NO	Public
FOREST REHAB	Evaluation of new forestry practices in North America: does forest management rehabilitat and maintain important ecological processes and structures?	10/12/2005	09/09/2007	NF	NA	NO	NO	Public
FORESTA	FORest conservation and EcoSysTem Accounting. Towards the	01/05/2014	30/04/2016	NF	NA	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	integration of private and public values into land use decisions modeling at farm scale. An application to Andalusia montes							
FORESTERRA	Enhancing FOrest RESearch in the MediTERRAnean through improved coordination and integration	01/01/2012	31/12/2015	http://www.foresterra.eu/	Not specifically	YES	NO	Public
FORLIVE	Forest management by small farmers in the Amazon - an opportunity to enhance forest ecosystem stability and rural livelihood	01/02/2005	31/01/2009	NF	NA	NO	NO	Public
FORMOD	Forest Models for Sustainable Forest	01/09/1999	31/08/2002	NF	NA	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	Management							
FORTHREATS	European Network on emerging diseases and threats through invasive alien species in forest ecosystems	01/02/2007	31/01/2009	NF	NA	YES	NO	Public
FUNDIVEUROPE	Functional significance of forest biodiversity in Europe	01/10/2010	31/03/2015	http://www.fundiveurope.eu/	Yes	YES	NO	Public
GEOLAND	Geoland - GMES products & services, integrating EO monitoring capacities, to support the implementation of European directives and policies related to "land cover and vegetation"	01/01/2004	31/03/2007	http://www.geoland2.eu/	Not specifically	YES	NO	Public
GLOCHAMORE	Global Change in Mountain	01/11/2003	31/10/2005	http://www.unesco.org/new/e	Partially	YES	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	Regions: An Integrated Assessment of Causes and Consequences			n/natural-sciences/environment/ecological-sciences/specific-ecosystems/mountains/glochamore/				
GNU	GMES network of users	01/10/2007	30/09/2010	http://www.fp7helm.eu/gnu/	Not specifically	YES	NO	Public
HERCULES	Sustainable futures for Europe's HERitage in CULtural landscapES: Tools for understanding, managing, and protecting landscape functions and values	01/12/2013	30/11/2016	http://www.hercules-landscapes.eu/	Not specifically	NO	NO	Public
IMECC	Infrastructure for Measurement of the European Carbon Cycle	01/04/2007	30/09/2011	http://imecc.ipsl.jussieu.fr/	Not specifically	YES	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
INCA-CO2	International Co-operation actions on CO2 capture and storage	01/10/2004	29/02/2008	NF	NA	YES	NO	Public
INNOVAWOOD SSA	An innovation strategy to integrate industry needs and research capability in the European forestry-wood chain	01/09/2005	29/02/2008	http://www.innovawood.com/	Not specifically	YES	NO	Public
INTEGRAL	Future-oriented integrated management of European forest landscapes	01/11/2011	31/10/2015	http://www.integral-project.eu/	Yes	YES	NO	Public
LEDDRA	Land and Ecosystem Degradation and Desertification: Assessing the Fit of Responses	01/04/2010	31/03/2014	http://leddra.aegean.gr/index.htm	Partially	YES	NO	Public
LINKTOFUN	Linking tree and belowground biodiversity to	01/03/2013	28/02/2017	NF	NA	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	forest Ecosystem function							
LITCOAST	Management of coastal forests of Lithuania: sustaining and enhancing forest health through silviculture	01/12/2006	30/11/2010	http://www.slu.se/en/departments/forest-mycology-plantpathology/research/forest_pathology/litcoast/	Partially	NO	NO	Public
MEDIGRID	Mediterranean Grid Of Multi-Risk Data And Models	01/11/2004	31/10/2006	http://www.medigrid.de/index_en.html	Not specifically	NO	NO	Public
MENFRI	Mediterranean Network of Forestry Research and Innovation (MENFRI)	01/12/2013	30/11/2016	http://www.etrera2020.eu/r21-clusters/11-med-cluster/11-mediterranean-network-of-forestry-research-and-innovation-menfri.html	Not specifically	YES	NO	private
MYCOIND	Mycorrhizas and Europe s oaks: a functional biodiversity	23/08/2010	22/08/2012	NF	NA	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	knowledge gap							
NA	Biological criteria for sustained development in natural degenerate forests of mediterranean Europe	01/03/1991	31/08/1993	NF	NA	NO	NO	Public
NA	Early response areas for climate change in Eurasia - Spatio-temporal dynamics of upper tree line in the Ural Mountains and implications for carbon sequestration	01/05/2002	30/04/2005	NF	NA	NO	NO	Public
NORTH STATE	Enabling Intelligent GMES Services for Carbon and Water Balance Modeling of Northern Forest	NA	NA	http://www.northstatefp7.eu/	Partially	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	Ecosystems							
OPERAS	Operational Potential of Ecosystem Research Applications	01/12/2012	30/11/2017	http://www.operas-project.eu/	Yes	NO	NO	Public
ORCHESTRA	Open architecture and spatial data infrastructure for risk management	01/09/2004	29/02/2008	http://www.eu-orchestra.org/	Not specifically	NO	NO	Public
PALMS	Palm harvest impacts in tropical forests	01/01/2009	31/12/2013	http://www.fp7-palms.org/	Not specifically	NO	NO	Public
PASTFORWARD	Development trajectories of temperate forest plant communities under global change: combining hindsight and forecasting (PASTFORWARD)	01/06/2014	31/05/2019	NF	NA	NO	NO	Public
POPFACE	Effects of atmospheric	01/05/1998	31/10/2001	NF	NA	YES	YES	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	carbon enrichment of cultivated terrestrial ecosystems: a face experiment on short rotation intensive polar plantation							
PORT CHECK	Development of generic 'on site' molecular diagnostics for eu quarantine pests and pathogens'	01/03/2004	31/08/2007	NF	NA	NO	NO	Public
PREFER	Space-based Information Support for Prevention and REcovery of Forest Fires Emergency in the MediteRranean Area	01/12/2012	30/11/2015	http://www.prefer-copernicus.eu/index.php/project-description	Not specifically	NO	NO	Public
PROFOR	Protected Forest Areas	28/03/2001	28/02/2006	NF	NA	NA	NA	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
PUMPSEA	Peri-urban mangroves forests as filters and potential phytoremediators of domestic sewage in East Africa	01/02/2005	31/07/2008	http://www.pumpsea.icat.fc.ul.pt/main.php	Not specifically	YES	NO	Public
RAPRA	Risk analysis for Phytophthora ramorum, a newly recognised pathogen threat to Europe and the cause of Sudden Oak Death in the USA	01/01/2004	31/03/2007	http://rapra.fera.defra.gov.uk/	Not specifically	NO	NO	Public
REAL	Resilience in East African Landscapes: Identifying critical thresholds and sustainable trajectories – past, present and future	01/09/2013	31/08/2017	http://www.real-project.eu/	Partially	NO	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
RESTORE	Resilience and stability in developing tools for sustainable forest management and restoration	01/03/2009	28/02/2011	NF	NA	NO	NO	Public
RISK-BASE	Coordination Action on Risk Based Management of River Basins	01/09/2006	31/12/2009	NF	NA	NO	NO	Public
ROBIN	Role Of Biodiversity In climate change mitigatioN	01/11/2011	31/10/2015	http://robinproject.info/home/	Yes	NO	NO	Public
SAGE	Simulating adaptation of forest management to changing climate and disturbance regimes	01/04/2013	30/09/2016	NF	NA	NO	NO	Public
SUMFOREST	Tackling the challenges in sustainable and multifunctional forestry through enhanced	01/01/2014	31/12/2017	http://era-platform.eu/era-nets/sumforest/	Partially	YES	NO	Public

Project acronym	Full project title	Start date (NA=Not Available)	End date (NA=Not Available)	Web page (NF=Not Found)	Linkage to biodiversity and ecosystem services (forest ecosystems included)	Italy as partner in the Project Consortium	Italy as coordinator of the Project Consortium	Type of partnership from Italy
	research coordination for policy decisions							
TEEMBIO	Toward Eco-Evolutionary Models for BIODiversity Scenarios	01/01/2012	31/12/2016	NF	NA	NO	NO	Public
TRANZFOR	Transferring research between EU and Australia-New Zealand on forestry and climate change	01/02/2009	30/06/2013	http://www.tranzfor.eu/	Not specifically	NO	NO	Public
WARECALC	Water resources vulnerability to climate and anthropogenic landscape changes	15/04/2009	14/04/2013	NF	NA	YES	YES	Public
WARM	Wildland-urban area fire risk management	01/12/2001	31/05/2004	NF	NA	NO	NO	Public

Appendix 2

Questionnaire structure and methodology

Section 1 – General description of the area

- Official name of the Park
- Name of the Management Authority
- Location (Municipalities, Provinces, and Regions included)
- Total area (hectares)
- Total forest area (hectares)
- Area not under forest management (hectares)
- Main Forest Categories
- Contact details

Section 2 - Forest Ecosystem Services relevance

This section is aimed at assessing FES relevance in the area. The assessment is carried out by assigning a ranking value to each specific service Class, according to the framework proposed by CICES V4.3 (http://cices.eu/; Haines-Young and Potschin 2013). For the purposes of this research, correspondences between CICES V4.3 Classes and FES Types have been created. Ranking values vary as follows: 0 (not important), 1 (less important), 2 (averagely important), 3 (very important), 4 (primary, fundamental). While ranking the FES classes, only forest ecosystems have to be considered (with the exception of cultural/aesthetic services, which may be considered at a broader scale).

CICES V4.3 framework				Related FES	FES Relevance
Section	Division	Group	Class	FES Type	
Provisioning	Nutrition	Biomass	Wild plants, algae and their outputs	Non-wood forest products availability	
		Water	Surface water for drinking	Fresh water availability	
			Ground water for drinking		
	Materials	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing	Wood mobilization and timber extraction	

CICES V4.3 framework				Related FES	FES Relevance
Section	Division	Group	Class	FES Type	
				(production of raw materials)	
	Energy	Biomass-based energy sources	Plant-based resources	Wood mobilization and timber extraction (for energy supply)	
Regulation and Maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals	Bioremediation	
			Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals		
Regulation and Maintenance	Mediation of flows	Mass flows	Mass stabilisation and control of erosion rates	Hydrogeological protection	
			Buffering and attenuation of mass flows		
		Liquid flows	Hydrological cycle and water flow maintenance		
			Flood protection		
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection	Pollination and seed dispersal	Biodiversity conservation	
			Maintaining nursery populations and habitats		
		Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations	Climate change mitigation	
			Micro and regional climate regulation		
Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes in different environmental settings	Improvement of tourism and recreation concerns	
			Physical use of land-/seascapes in different environmental settings		
		Intellectual and representative interactions	Scientific		
			Educational		
			Heritage, cultural		
			Entertainment		
			Aesthetic		
	Spiritual, symbolic and other interactions with biota, ecosystems, and	Spiritual and/or emblematic	Symbolic	Conservation of the landscape identity	
			Sacred and/or religious		
		Other cultural	Existence		

CICES V4.3 framework				Related FES	FES Relevance
Section	Division	Group	Class	FES Type	
	land-/seascapes	outputs	Bequest		

Calculation

A unique relevance value (REL) of each ecosystem service Division and for all of respondents was calculated by using the following equation:

$$REL = \frac{\sum_{i=1}^{m}\left[\sum_{j=1}^{n}(REL_j)\right]_i}{m} \; [0 \div 4]$$

Where: REL is the relevance value for the ecosystem services Division (e.g. nutrition, materials, energy, etc.); REL_j is the relevance value for the *j*th ecosystem service Group; n is the total number of ecosystem services Groups; and m is the total number of respondents.

Section 3 – Relationship between local stakeholders and Forest Ecosystem Services

This section is aimed at assessing how much local stakeholders (grouped by typology) currently influence the FES provision in the area. In a cross-table, several stakeholders are compared with each service Division (see CICES V4.3, cices.eu/). The assessment is carried out per each stakeholder/Division cross-section by assigning one of the following symbols: +1 (if the stakeholder is considered as a driver improving the related-service provision), -1 (if the stakeholder is considered as a barrier limiting the related-service provision), or 0 (if the stakeholder has no direct influence on the related-service provision).

		Nutrition	Materials	Energy	Mediation of waste, toxics and other nuisances	Mediation of flows	Maintenance of physical, chemical, biological conditions	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes
Stakeholder typology	Ecosystem services Division code	A	B	C	D	E	F	G	H

		Nutrition	Materials	Energy	Mediation of waste, toxics and other nuisances	Mediation of flows	Maintenance of physical, chemical, biological conditions	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes
Stakeholder typology	Ecosystem services Division code	A	B	C	D	E	F	G	H
Nature conservation	Non-governmental organizations								
	Protected area officials								
	European Union (e.g. NATURA2000 Network sites)								
	State managed national parks (for national environmental and cultural heritage)								
Agriculture	Large-scale farming								
	Pastoralism (e.g. sheep, reindeer)								
	Small-scale farming								
Tourism sector	Skiing resort businesses and workers								
	Nature-based tourism entrepreneurs								
	Rural tourism entrepreneurs (e.g. small scale bed and breakfast)								
	National and international tour operators								
Forestry sector	State forestry institutions								

		Nutrition	Materials	Energy	Mediation of waste, toxics and other nuisances	Mediation of flows	Maintenance of physical, chemical, biological conditions	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes
Stakeholder typology	Ecosystem services Division code	A	B	C	D	E	F	G	H
	Small-scale private forest owners								
	Private forest companies (industrial forestry corporations)								
Recreation activities	Hunters								
	Mushroom, berry or other non-wood products pickers								
	Recreationists (outdoor activities such as mountain bike cycling)								
	Skiers (general)								
	Users of snow mobiles, all-terrain vehicles, or other motorized ways to access and enjoy treeline area								
Education and research	Researchers, technicians and scientists (e.g. having study areas and sampling sites in the area)								
	Schools or other groups of people doing educational trips to the								

		Nutrition	Materials	Energy	Mediation of waste, toxics and other nuisances	Mediation of flows	Maintenance of physical, chemical, biological conditions	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes
Stakeholder typology	Ecosystem services Division code	A	B	C	D	E	F	G	H
	area								
Public Institutions	Army (strategic purposes and national security)								
Users and consumers	Residents using fresh water for drinking								
	Local farmers using water for agriculture purposes								
Local inhabitants	Permanent residents								
	Second home residents								
	Land owners								
Manufacturing sector	Mining companies								
	Green-power companies (wind, water, etc.)								

Calculation

A unique influence value (*INF*) was calculated for each Stakeholder typology and ES Division between all of respondents by using the following equation:

$$INF = \frac{\sum_{i=1}^{m}(INF_i)}{m}\ [-1 \div 1]$$

Where: INF_i is the influence value on the *i*th ecosystem services Division; and m is the total number of respondents.

Section 4 – Governance instruments at work in the area

This section aims at identify what are the governance instruments currently at work in the broader area (i.e. in and outside the Park boundaries). Answers are in the TRUE/FALSE form.

Governance scale	Governance instrument	Currently at work
Large-scale urban planning	(Strategic) Regional and sub-regional land-use plan	
	Main Regulatory Plan	
	Plan for productive settlements (Municipalities land use plan)	
Forest sector planning	Management and Conservation Plan (Protected Area management plan)	
	Regional Forest Plan	
	Watershed Plan	
	Forest Landscape Management Plan	
	Regional Forest Law	
	EU regulatory frameworks (CAP, NATURA2000 Network, etc.)	
	National Strategies and Forest Action Plans	
Market-based governance	Eco-labels for local agriculture products (tourism purposes)	

Calculation

Answers were converted into numeric values, as follows: TRUE in 1, and FALSE in 0.

Section 5 – Factors considered in decision-making processes

In this section, a list of sentences on how different factors (e.g. local stakeholders involvement, FES assessment, trans-disciplinarity, etc.) have an active influence (or are considered) within decision-making processes for forest management in the Park is provided. Answers are in the TRUE/PARTIALLY TRUE/FALSE form.

The relevant stakeholders have the possibility to participate in the decision-making processes
The stakeholders can participate in land use planning processes with a confidence that their view is properly taken into account
Governance instruments work in balanced combination of bottom-up and top-down practices
Governance instruments are transparent and include a continuous knowledge transfer between stakeholders, Park managers, and Public Bodies and Institutions

There are disagreements and disapprovals about currently implemented forest management practices among citizens
Local stakeholders know forest managers and how to reach them in cases of direct involvement in decision-making processes
The decision-making processes in forest governance frequently use technological and scientific progresses currently available both at national and international level
European and national conservation guidelines in forest governance are actually taken into account during decision-making processes, as well as they are implemented in forest management plans
The ecologic and economic evaluations of Forest ecosystem services are currently considered during decision-making processes
Current forest management derives from the analysis of ecosystem service trade-offs

Calculation

Answers were converted into numeric values, as follows: TRUE in 1, PARTIALLY TRUE in 0.5, and FALSE in 0.

Section 6 – Decision-making processes and research activities

In this section, a list of sentences about the role of science and research in decision-making processes for forest management in the Park is provided. Answers are in the TRUE/PARTIALLY TRUE/FALSE form.

Researchers and scientists from different fields (e.g. environmental sciences) are supported by the Management Authority of the Park during their activities within the area
The Management Authority has well-established contacts with Public or private Research Institutions at least at national level
Research projects include continuous two-way knowledge exchange between researchers and stakeholders (e.g. local stakeholders, decision makers)
Research results/outcomes (and advances) are conceived by the Management Authority of the Park as fundamental in supporting and completing the traditional knowledge and techniques in forest management and planning
The Management Authority of the Park actively participate in Research project at national or European level

Calculation

Answers were converted into numeric values, as follows: TRUE in 1, PARTIALLY TRUE in 0.5, and FALSE in 0.

Section 7 – Forest Ecosystem Services relevance in forest management

In this section, a test is carried out in order to assess the relevance of FES while preparing a Forest Management Plan for the next 30 years in the Park area. The following three are the assessment elements: (i) priority in forest management; (ii) difficulty in valuing/quantifying the service; (iii) relevance for local communities in terms of expected benefits. The first elements is assessed by using ranking values from 0 (low priority) to 5 (high priority). The second element is assessed by using ranking values from 0 (low difficulty) to 5 (high difficulty). The third element is assessed by using ranking values from 0 (low relevance) to 5 (high relevance). In case of a recently implemented Forest Management Plan, ranking values are properly derived from it.

Forest Ecosystem Service (FES)	Priority in forest management	Difficulty in valuing/quantifying the service	Relevance for local communities in terms of expected benefits
Wood mobilisation and timber extraction			
Non-wood forest products availability			
Fresh water availabilty			
Hydrogeological protection (or against other natural extreme events)			
Biodiversity conservation (habitat integrity and diversity, genepools protection, etc.)			
Climate change mitigation			
Bioremediation			
Conservation of the landscape identity (cultural, spiritual, and aesthetic)			
Improvement of tourism and recreation concerns			

Calculation

The influence value of ecosystem services in Forest Management (*FESFM*) was calculated for each of the three elements (priority, difficulty, and relevance) and for each FES Division by using the following equation:

$$FESFM = \frac{\sum_{i=1}^{m}(FESFM_i)}{m} \quad [0 \div 5]$$

Where: $FESFM_i$ is the influence value of the ith FES in forest management; and m is the total number of respondents.

References

Haines-Young, R. and Potschin, M. 2013 CICES V4.3 – Revised report prepared following consultation on CICES Version 4, August-December 2012. The University of Nottingham, Centre for Environmental Management, p. 32.

Appendix 3

Details of forestry interventions

Table 3.1: Specifications about Forest Management Strategies, as adopted for each Forest Category with Regular Management approach.

FC	FMS	Description	Frequency (intervening year)	Intensity (% of the total biomass)
Montane beech forest	Even-aged high forest	Shelterwood system • Rotation period: 110 years • Seed cut: 110 years (40% biomass removed) • Secondary cut: 120 years (55% biomass removed) • Final cut: 130 years (85% biomass removed) • Thinnings: 15-year selective thinning until 65 years, then after 25 years until year 90. In any case, selective thinning is adopted to progressively reduce the stand biomass (from 30% to 15% biomass removed), while maintaining the growing capacity of remaining trees • Regeneration: natural regeneration • Biomass residual: 30-35 Mg	15 30 45 65 90 110 120 130 145 160 175 195 220 240 250 260 275 290	0.3 0.2 0.15 0.15 0.15 0.4 0.55 0.85 0.3 0.2 0.15 0.15 0.15 0.4 0.55 0.85 0.3 0.2
Montane beech forest	‘High-coppice’ forest	Conversion to high-forest through the following steps: • Natural development of the ageing coppice forest. No intervention is planned over the first 80 years • Minimum stock: 25-30 m^3 (standards release at the beginning of simulation period) • Seed cut at year 80: 40% biomass removed • Secondary cut at year 90: 55% biomass removed • Final cut at year 100: 85% biomass removed	80 90 100 115 130 145 160 175	0.4 0.55 0.85 0.3 0.25 0.2 0.15 0.15

FC	FMS	Description	Frequency (intervening year)	Intensity (% of the total biomass)
		• Shelterwood system for the established high-forest • Rotation period: 110 years • Thinnings: 15-year selective thinning until 190 years (from 30% to 15% biomass removed) • Seed cut at year 210: 40% biomass removed • Secondary cut at year 220: 55% biomass removed • Final cut at year 230: 85% biomass removed	190 210 220 230 245 260 275 290	0.15 0.4 0.55 0.85 0.3 0.25 0.2 0.15
Mediterranean and Anatolian fir forest	Even-aged high forest	Clearcutting system • Rotation period: 100 years (to avoid root rot phenomena) • Thinnings: 10-year moderate thinning from below (from 15% to 7% biomass removed) • Regeneration: seeding/planting • Minimum stock: none	20 30 40 50 60 70 80 100 120 130 140 150 160 170 180 200 220 230 240 250 260 270 280 300	0.15 0.15 0.1 0.1 0.1 0.07 0.07 1 0.15 0.15 0.1 0.1 0.1 0.07 0.07 1 0.15 0.15 0.1 0.1 0.1 0.07 0.07 1
Mediterranean and Anatolian	Even-aged high	Clearcutting system	20	0.3

FC	FMS	Description	Frequency (intervening year)	Intensity (% of the total biomass)
black pine forest	forest	• Rotation period: 100 years • Thinnings: 20-year moderate thinning from below (from 30 to 15% biomass removed) • Regeneration: natural regeneration • Minimum stock: none	40 60 80 100 120 140 160 180 200 220 240 260 280 300	0.25 0.15 0.15 1 0.3 0.25 0.15 0.15 1 0.3 0.25 0.15 0.15 1
Thermophilous pine forest	Even-aged high forest	Clearcutting system • Rotation period: 65 years • Thinnings: progressive thinning from below (5 to 25-year, from 35% to 12% biomass removed) • Regeneration: planting • Minimum stock: none	5 10 25 45 65 70 75 90 110 130 135 140 155 175 195 200 205 220 240	0.35 0.3 0.2 0.12 1 0.35 0.3 0.2 0.12 1 0.35 0.3 0.2 0.12 1 0.35 0.3 0.2 0.12

FC	FMS	Description	Frequency (intervening year)	Intensity (% of the total biomass)
			260 265 270 285	1 0.35 0.3 0.2
Subalpine and mountainous spruce and mountainous spruce-silver fir mixed forest	Uneven-aged high forest	Selection system: • Cutting period: 15 years • Thinnings: moderate selective thinning (20% biomass removed) • Minimum stock: 200-250 m^3	15 30 45 60 75 90 105 120 135 150 165 180 195 210 225 240 255 270 285 300	0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2
Mediterranean evergreen oak	‘High-coppice’ forest, then even-aged forest	Conversion to high-forest through the following steps: • Natural development of the ageing coppice forest. No intervention is planned over the first 80 years • Minimum stock: 5-10 m^3 (standards release at the beginning of simulation period) • Seed cut at year 80: 40% biomass removed • Secondary cut at year 90: 55% biomass removed • Final cut at year 100: 85% biomass removed	80 90 100 115 130 145 160 175 190	0.4 0.55 0.85 0.3 0.25 0.2 0.15 0.15 0.15

FC	FMS	Description	Frequency (intervening year)	Intensity (% of the total biomass)
		• Shelterwood system for the established high-forest • Rotation period: 110 years • Thinnings: 15-year selective thinning until 190 years (from 30% to 15% biomass removed) • Seed cut at year 210: 40% biomass removed • Secondary cut at year 220: 55% biomass removed • Final cut at year 230: 85% biomass removed	210 220 230 245 260 275 290	0.4 0.55 0.85 0.3 0.25 0.2 0.15

Table 3.2: Specifications about Forest Management Strategies, as adopted for Montane beech forest in CAP and ASI with Alternative Management approach.

Case study	FC	FMS	Description	Description	Frequency (intervening year)	Intensity (% of the total biomass)
CAP	Montane beech forest	'High-coppice' forest	Conversion to high-forest through the following steps: • Natural development of the ageing coppice forest. No intervention is planned over the first 80 years • Minimum stock: 5-15 m^3 (standards release at the beginning of simulation period) • Seed cut at year 80: 40% biomass removed • Secondary cut at year 90: 55% biomass removed • Final cut at year 100: 85% biomass removed • Selection system for the established high-forest • Thinnings: 15-year selective thinning (20% biomass removed)		80 90 100 115 130 145 160 175 190 205 220 235 250 265 280 295	0.4 0.55 0.85 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2 0.2
ASI	Montane beech forest	Uneven-aged high forest	Selection system: • Cutting period: 15 years Thinnings: moderate selective thinning (15% biomass removed)		15 30 45	0.15 0.15 0.15

Case study	FC	FMS	Description	Description	Frequency (intervening year)	Intensity (% of the total biomass)
			• Minimum stock: 200-250 m^3		60 75 90 105 120 135 150 165 180 195 210 225 240 255 270 285 300	0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15 0.15

Special thanks

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